An assessment of the relationship between habitat controls and Atlantic salmon and brown trout abundance using remote sensing and GIS in the river Eden catchment

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Abstract

A new approach to the assessment of relationships between habitat controlling processes and salmon and trout abundance is presented and applied to the River Eden, Cumbria, UK. The potential of Geographical Information Systems (GIS), remote sensing, aerial photography, risk-based environmental modelling and electrofishing is demonstrated for the collection and integration of habitat and species abundance data at the scale of large catchments (>1000km^2). Based on this data, a key output of the research is the development of a spatially-structured, hierarchical database that allows hypotheses regarding the relationship between habitat controls and salmon and trout abundance to be tested at multiple scales. In particular, assessment has been made at the whole catchment-scale (2,300km^2) and then at a series of sub-catchment scales (10-100s km^2). Analyses at these two scales revealed contrasting results, emphasising that the scale of observation and analysis is crucial in determining the relationships identified. In the catchment-scale analysis, salmon and trout abundance were significantly correlated with the catchment-scale process of surface hydrological connectivity, both weighted and un-weighted by land cover. However, as the scale of analysis contracted, the spatial variance exhibited by catchment-scale processes declined and more local-scale riparian and in-stream habitat controlling processes such as cover and bank erosion became important. These results provide evidence in support of theories which suggest a hierarchical structuring of catchments where large scale processes provide the structure within which riparian and in-stream habitat controls operate. Results are also presented showing that fish abundance responds and maps onto to this hierarchical structuring in different ways depending on the potential for mobility at different life-stages and the location of habitat utilised within the landscape.

Based on these results it is concluded that effective habitat restoration strategies must adopt a multi-scale approach in which in-stream and riparian scale actions are situated within the context of their controlling catchment-scale processes. The concept of hydrological connectivity is also recommended as an effective tool by which to assess the influence of landscape factors such as land cover on in-stream condition and salmon and trout abundance.

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Data	Developer	Licence and purchase agreements
OS Land-Line	Ordnance Survey	Environment Agency, 100026380 (2007). Schedule 5 contractor licence between Environment Agency and Compass Informatics
OS 1:50,000 raster	Ordnance Survey	Environment Agency, 100026380 (2007). Schedule 5 contractor licence between Environment Agency and Compass Informatics
NEXTMap Great Britain 5m DTM	Intermap Technologies Inc	Perpetual licence purchased by Eden Rivers Trust through Getmapping PLC December 2003.
Land Cover Map 2000	Centre for Ecology and Hydrology	Environment Agency sub-contractors agreement with Eden Rivers Trust 01/12/2003 – 01/12/2006
1961-2000 baseline, 5km gridded dataset of mean annual rainfall	UK Climate Impacts Programme & UK Meteorological Office	Licensed to Prof. S. Lane and Dr. S. Reaney of Durham University for use in the development and application of the *SCIMAP* model
20cm resolution digital aerial photography of the River Eden and tributaries	Compass Informatics	Purchased by Eden Rivers Trust (2004)
River centreline digitised form 1:50,000 OS raster data	CHASM, Newcastle University	Downloaded freely from CHASM web-site November 2003.
Salmonid population data	Eden Rivers Trust Environment Agency	Provided freely by Eden Rivers Trust and the Environment Agency with many thanks to APEM Ltd, Atlantic Salmon Trust and Natural England for their support
SCIMAP (The Sensitive Catchment Integrated Modelling Platform)	Durham and Lancaster Universities	Provided freely

Declaration of copyright

I confirm that no part of the material presented in this thesis has previously been submitted by me or any other person for a degree in this or any other university. In all cases, where it is relevant, material from the work of others has been acknowledged.

Signed:..

Chapter One - Introduction

1.1 Research aims and objectives

The aim of this research is to couple recent advances in remote sensing, Geographical Information Systems (GIS), environmental modelling and ecological surveying techniques with current ecological understanding of habitat controls on salmonid populations, to help develop a more effective approach to prioritising habitat restoration. This will be a generic study but developed and applied to the River Eden catchment, Cumbria, UK, in particular focusing on the salmonid species, *Salmo salar* (Atlantic salmon) and *Salmo trutta* (brown trout). The aim of the research will be achieved through the following objectives:

Objective (1): To review and to synthesise current understanding of in-stream, riparian and catchment-scale controls on freshwater salmonid habitat throughout the lifecycle of salmon and trout, to help formulate a set of hypotheses for further investigation.

Objective (2): To employ recent advances in remote sensing, GIS and environmental modelling, to identify, to develop and to validate tools for quantifying the habitat of salmon and trout at the catchment-scale, appropriate to each habitat control and scale of control.

Objective (3): To use the data acquired under (2), to investigate hypotheses formulated through (1) regarding habitat controls and salmonid populations, and to discuss the results in the context of effective approaches to habitat restoration.

The aim of this chapter is to present an overview and introduction to the research. This will first consider the wider context within which the thesis is set, outlining the need for the research (1.2). Second, the specific case study approach will be explained, including an introduction to the River Eden catchment (1.3). Third, each of the above thesis objectives will be discussed in detail, highlighting their contribution to the overall aim (1.4). The chapter will conclude by presenting the thesis structure (1.5).

1.2 Research context

1.2.1 Historic approach to fisheries management and habitat restoration

Many salmonid fisheries around the world have been reported to be in decline over recent years (Figure 1.1), with numerous theories for this decline proposed and discussed in the scientific literature. These include, climate change (e.g. Swansberg *et al.*, 2002; Solomon and Sambrook, 2004; and Boylan and Adams, 2006), disease (e.g. Peeler *et al.*, 2004; and Roberts, 1993) exploitation (e.g. Bowker *et al.*, 1998; and Almodova and Nicola, 2004), competition from invasive species (e.g. Griffiths *et al.*, 2004), aquaculture (e.g. Gross, 1998; and Read and Fernandes, 2003) and freshwater habitat degradation (e.g. O'Grady and Duff, 2000; Hendry *et al.*, 2003). In particular, the last of these theories has received considerable attention from both researchers and practitioners over the years, and there has been a series of international workshops dedicated to salmonid habitat enhancement, which have run since 1978 (Duff, 2002). Whilst it is likely that salmonid population decline is not attributable to habitat degradation alone, but rather to the complex interaction of all of the above factors, the prominence given to habitat may, in some respects, reflect the fact that it is actually something fisheries managers can visibly see, control, and have influence over. In other words, it is the area where fisheries managers feel their actions are most likely to make the greatest difference to stocks.

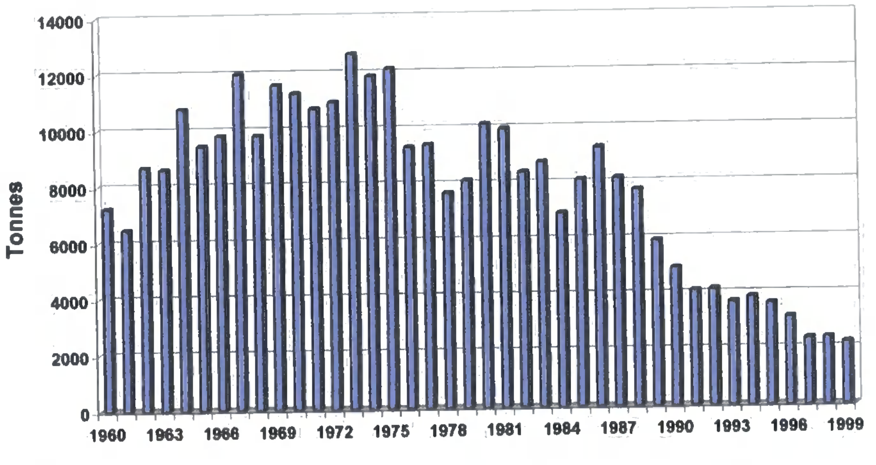

Figure 1.1: *Decline in the global nominal Atlantic salmon catch since 1960.* ***(Source:*** *Salmonid 21C, www.salmonid21C.org).*

This premise has resulted in a long history of freshwater habitat improvement schemes, beginning in North America in the 1930s (Duff, 2002). However, to date, restoration has often been localised, short-term and based on subjective assessments, directed by funding opportunities (Folt *et al.*, 1998). Whilst such projects can have beneficial effects at a local level, the widespread improvement in populations that is desired is not always achieved. Localised restoration projects may be unsuccessful for the following reasons:

(1) They frequently fail to capture a fundamental biological property of salmonid populations: their life-stage dependent habitat requirements and associated mobility throughout the river system.

(2) They often focus solely on local physical habitat conditions and fail to consider habitat controls which operate at a range of different spatial scales, often remote, but nevertheless connected to, and impacting on, the site in question.

(3) They often apply findings from research undertaken at one spatial scale or location to management undertaken at another, without accounting for differences in the spatial heterogeneity of habitat controls across different scales and in different locations.

To address these issues, there is a need to understand the relationships between habitat controls and salmonid populations across a wider range of spatial scales than is possible from local analysis. It is readily acknowledged that habitat availability and quality exert significant influence over population dynamics both by limiting carrying capacity and stimulating the operation of density-dependent effects such as territorial competition and food availability, and through density-independent effects such as the siltation of spawning gravels (Milner *et al.*, 2003). We know much less about what factors control habitat availability and quality, the scale at which they operate, and the life-stage they affect.

1.2.2 The need to consider species mobility

It is frequently recognised within the scientific literature that scale is an important factor to consider when undertaking investigations in fisheries science (e.g. Folt *et al.*, 1998; Stauffer *et al.*, 2000; Armstrong *et al.*, 1998; Pess *et al.*, 2002; Wang *et al.*, 2003). This is particularly true when examining the relationship between habitat and highly mobile species such as the Atlantic salmon and brown trout. As Folt *et al.* (1998, p.9) state: "Major shifts in behaviour and habitat use over ontogeny, along with a relatively long life span and large dispersal and migration distances, make scale issues critical for effective conservation, management, and restoration of Atlantic salmon". The same statement could equally be applied to the case of brown trout. Figure 1.2 presents a schematic representation of the life-cycle of the Atlantic salmon and brown trout whilst Table 1.1 summarises life-stage dependent habitat requirements. It is important to note that the classification of habitat requirements is far from clear cut with considerable overlap of ranges between species and between and within life-stages (Armstrong *et al.*, 2003). However, it is clearly evident that salmonids make extensive use of the river environment during their life-cycle, moving throughout the catchment and occupying different scales of habitat, in different locations, at different times. Consequently, successful and sustainable management may only be achieved through restoration that considers all these habitats, and hence, the catchment-scale. For example, localised restoration increasing spawning potential and juvenile numbers at one site may subsequently be constrained at a later stage in the life-cycle, for example, by a lack of over-wintering habitat (Armstrong *et al.* 2003). Conversely, understanding a decline in adult populations of brown trout or returning Atlantic salmon in the main stem of a river may require understanding of juvenile (fry and parr) production and hence juvenile habitat availability in many tributaries across a much wider spatial extent.

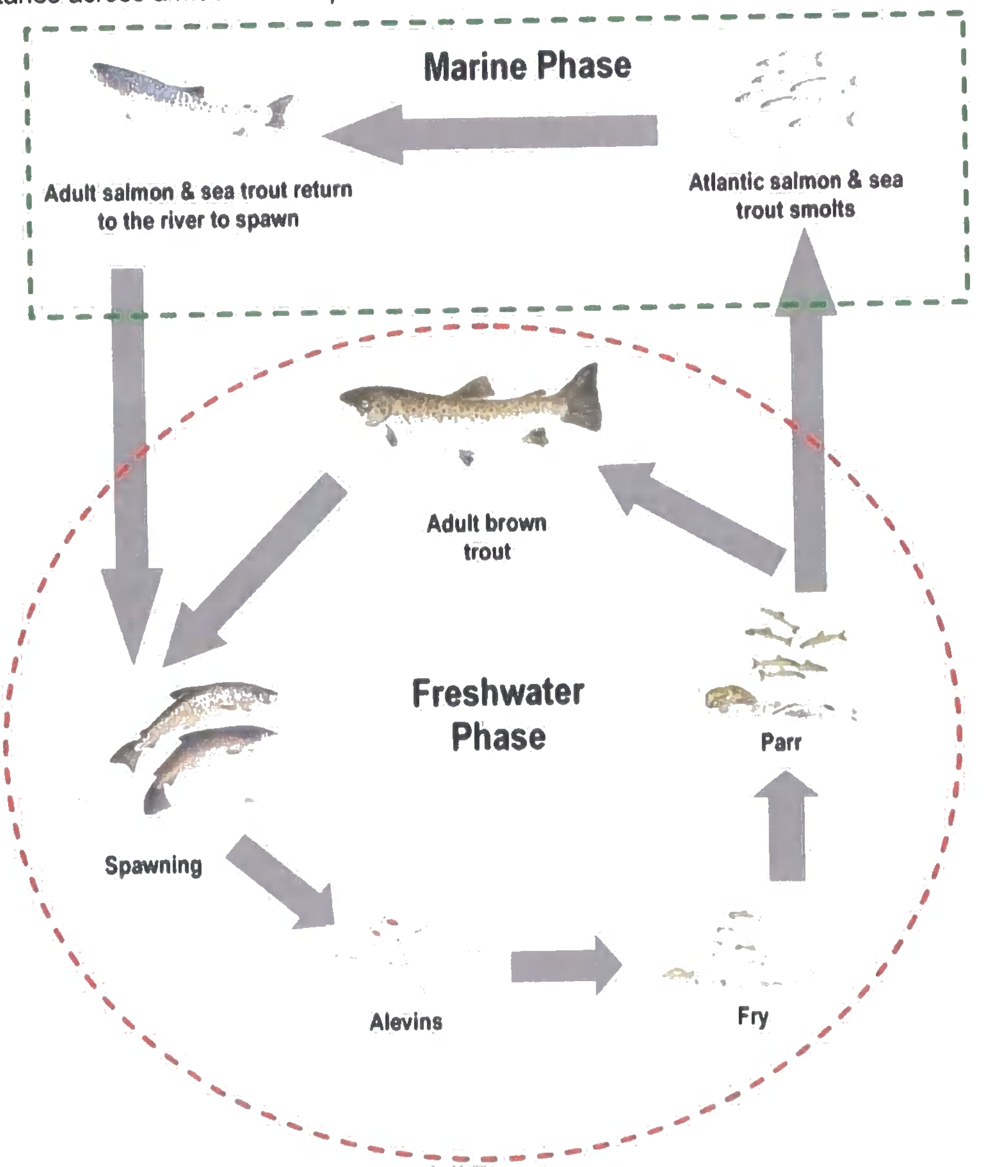

Figure 1.2: *The salmonid life-cycle (artwork courtesy of B.Bewick)*

Table 1.1: *A summary of salmonid life-stage dependent habitat requirements. Based upon Mills, (1989), Armstrong et al., (2003); Crisp, (1996); Hendry and Cragg-Hine, (2003); and Armstrong et al., (1998).*

Life-stage	General Description	Atlantic salmon	Brown trout
Spawning	Eggs are deposited in a series of nests 'redds' that are excavated into the gravel and then covered. A good supply of dissolved oxygen is essential for survival to emergence; hence, riffles that are free from fines are the preferred habitat. Salmonids typically spawn between October and December. Within any given catchment trout tend to spawn earlier than salmon and make more use of small headwaters.	Water velocity: 35-80 cms^{-1} Water depth: 17-76 cm Substrate size: 20-30 mm Burial depth: 13-30 cm	Water velocity: 10-80 cms^{-1} Water depth: 6-82 cm Substrate size: 8-128mm Burial depth: 8-25 cm
Alevins	On hatching fish are called alevins. They remain in the gravel at their spawning site feeding on their yolk sacs. They require a good supply of dissolved oxygen for survival. **Spatial range: 0-10 cm**	As for spawning	As for spawning
Fry	By the time the fish emerge from the gravel the yolk sac has been absorbed: they are ready to start actively feeding and are known as fry. It is thought that dispersal of most fry from the redd is very limited in extent during the first summer of life. As such, density-dependent effects are particularly high during this period. Riffles are typical fry habitat. **Spatial range: 1-100s m**	Water velocity: 10-40 cms^{-1} Water depth: 20-40 cm Substrate size: 16-256 mm	Water velocity: 0-20 cms^{-1} Water depth: 5-35 cm Substrate size: 10-90 mm
Parr	Following dispersal from the redd and nursery areas during autumn and winter fish are reffered to as parr. At this time they develop dark patches along their sides known as parr marks and opt for deeper water as they grow. Trout show a preference for slightly deeper water than salmon parr which can hold station with less expenditure of energy. Overhead cover is very important for trout. Therefore, trout may dominate in narrow streams where bankside shelter is abundant relative to the stream bed, whilst salmon may dominate in the faster flower sections of wider streams where they are free from competition. Juveniles are termed parr until smoltification of salmon and sea trout or maturation of adult brown trout. **Spatial range: 0.01-10 km**	Water velocity: 10-65 cms^{-1} Water depth: 20-70 cm Substrate size: 64-512+ mm	Water velocity: 10-70 cms^{-1} Water depth: 14-122 cm Substrate size: 8-128 mm
Smolt	Smoltification of salmon and sea trout involves physiological, morphological and behavioural changes, which generally occur between 1 and 4 years when parr reach 100–120 mm in length. Smolts are characterised by their silvery livery and exhibit shoaling behaviour as they migrate to sea typically between April-June. Most migration takes place at night and may be triggered by a high flow and change in water temperature. Cover is required in the form of boulders, overhung banks and deep pools as protection from predators. **Spatial range: 0.1 – 1000+ km**		
Adult brown trout	They remain in the freshwater environment and whilst some may remain within meters of their redd others may disperse considerable distances typically in a downstream direction to main stem rivers and lakes. Upon reaching sexual maturation, typically after 2-4 years, brown trout are termed adults. They migrate upstream to their tributary of origin during October - December to spawn, typically in small upland tributaries and can reproduce for several years. Adult trout are highly territorial and aggressive. In addition to invertebrates they will also feed on smaller fish including trout fry and parr. They typically occupy deep pools, requiring cover in the form of overhanging vegetation, tree roots, logs and undercut banks. **Spatial range 0.01- 1000 km**		
Adult salmon and sea trout	Upon reaching sexual maturation salmon and sea trout are termed adults and return to their natal river to spawn. Fish which return after one winter at sea are termed grilse. Fish that return after more than one winter at sea are termed multi-sea winter fish, commonly known as 'Spring run' fish when entering the river before June. On returning to the river to salmon and sea trout are believed not to actively feed. It is also thought that deep holding pools near to spawning areas may be important to provide cover as redd digging can take several days. A spent or spawned adult is termed a kelt. **Spatial range: 1000+ km**		

Additionally, factors that pose a problem for one life-stage may not impact others to any significant degree. For example, consider the issue of overshading. This is believed to limit Atlantic salmon populations by reducing autochthonous primary production and subsequently reducing the number of drift invertebrates which are an important food source (O'Grady, 1993). Overshading of adult habitat is considered less critical than overshading of juvenile (fry and parr) habitat, a variation that can be linked to different physiological requirements and habitat needs at different life-stages. Salmon fry and parr actively feed, primarily on invertebrates particularly aquatic insect larvae such as mayfly, caddis and stonefly. On migrating to sea, salmon undergo a period of rapid growth feeding primarily on other fish and crustecea (Hendry and Cragg-Hine, 1997). However, on returning to the river system to spawn, adult salmon are commonly believed not to feed (Mills, 1989). This onset of anorexia has been linked to fish reaching a threshold level in condition where they have acquired sufficient energy reserves for upstream migration and spawning (Kadri *et al.*, 1995). This is not always the case, and adult salmon have been caught with freshwater insects inside their stomach and gut. However, the number of insects found is generally low (Johansen, 2001). These variations in feeding habit mean salmon at different stages of their life-cycle will respond differently to the level of overshading. Restoration projects advocating the use of coppicing to reduce shade and encourage autochthonous production should be mindful of this. In fact, high levels of cover over pools may be required by adult salmon to provide protection from predators and bright sunlight, whilst resting between periods of active migration (Crisp, 1996). Whilst beneficial for juvenile life-stages, broad-scale coppicing aimed at improving populations could become detrimental to adult populations if targeted at habitat of the wrong life-stage.

Relationships with habitat may also be linked to the spatial range utilised by salmonids at different stages in their life-cycle (Figure 1.3). Fry have been observed to remain within 100s of metres of their spawning site. For example, Einum and Nislow (2005) observed fry to remain within 644m and 884m downstream, and 1,500m and 642m upstream of their redd in 2002 and 2003 respectively, with the median dispersal range being 92m and 41m in the two years. Similarly, Kennedy, (1982, *cited in* Crisp, 1996) found over 70% of fry to be within 100m downstream of their stocking point. There are always exceptions and Beall *et al.*, (1994, *cited in* Crisp, 1996) found that salmon had dispersed over 2000m downstream, with a substantial number moving between 1000m and 1500m by the October of their first year. It has been suggested that dispersal of fry is constrained by: (1) energetic costs and the lack of feeding opportunities during dispersal, which may lead to starvation; and (2) increased exposure to predators (Einum and

Nislow, 2005). This lack of mobility within the first few weeks and months of life means that fry are highly susceptible to density-dependent mortality and, as a result, to local habitat conditions which regulate the local carrying-capacity. Parr and adult populations can be more mobile, with dispersal ranging from 10s m-1000s m. Dispersal downstream is typically greater than the degree of dispersal upstream. Some dominant fish may aggressively defend a small localised territory which is profitable for food, whilst other, more subordinate fish may be more mobile and 'float' between territories (Suter and Huntingford, 2002). Some fish may remain within metres of their redd, whilst others migrate to main stem rivers, lakes or to sea (Elliott, 1994). This increased potential for mobility and dispersal results in mortality of parr and adult populations that tends to be density-independent, as they can move away from localised habitat pressures provided that the spatial extent of the pressure is less than the spatial range of their mobility. The opposite can also occur where localised degradation can affect the entire adult population of a catchment whilst having little effect on the juvenile population. For example, during the Atlantic salmon smolt run an entire population may be funnelled through one reach. If a pollution event were to occur there, the whole smolt population of that catchment could be wiped out, whereas only juveniles that inhabit that particular reach would be affected (Armstrong *et al.*, 1998). This poses an interesting question as to whether salmonid distributions exhibit relationships with habitat that are structured by life-stage according to the extent of their mobility at that stage in their life-cycle.

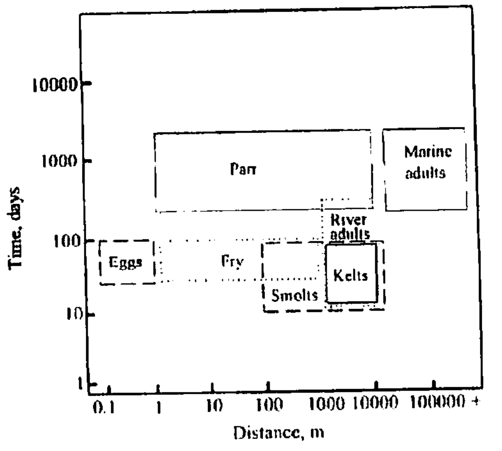

Figure 1.3: *Stages of the life-cycle of Atlantic salmon in relation to scales of space and time. (Armstrong et al., 1998)*

The complexities of the Atlantic salmon and brown trout life-cycle mean that it is essential to make connection between the scale and extent of degradation sources, the type of habitat impacted, and the life-stage this affects.

1.2.3 The need to consider the scale at which habitat controlling processes operate

It is not just fish which operate at different scales. The factors and processes which influence habitat quality at a point in the channel network also operate across a range of scales. These can be viewed within a hierarchical framework and include: (1) catchment-scale, processes such as the impact of topography, land use and hydrological connectivity on the delivery of diffuse pollution and water quality; (2) riparian scale processes such as the impact of bankside vegetation growth on shade and cover availability, and the impact of stock access, grazing pressure and bank erosion on siltation and channel morphology; and (3) at the in-stream scale, processes and variables such as depth, velocity, substrate size, gravel siltation and channel slope.

Historically, freshwater and fisheries management has focused on in-stream habitat conditions often undertaking restoration projects aimed at treating the symptoms of habitat degradation rather than the causes (Summers *et al.*, 1996). This has included strategies such as: the placement of in-stream structures to provide cover, increase habitat diversity and accumulate gravels (De Jong *et al.*, 1997; and O'Grady *et al.*, 2002); and gravel cleaning to alleviate the problems of siltation in spawning gravels (Shackle *et al.*, 1999). Such strategies can be highly successful. For example, O'Grady *et al.* (2002) noted a major increase in trout parr stock densities following a programme of in-stream enhancement works including log and stone weirs, boulder placement and pool creation in the Lough Ennell catchment, Ireland. However, unless the causes of habitat degradation are also dealt with, such strategies may fail to be sustainable in the long-term, without ongoing maintenance. In-stream structures may be damaged or washed downstream during high flows, whilst gravels may re-silt following cleaning. Instead, achieving sustainable restoration requires focus on the causes of habitat degradation. It is now widely recognised that processes which control in-stream habitat can operate outside the channel. In recognition of this restoration projects (e.g. livestock exclusion fencing) which address riparian scale processes are now commonplace and a central theme of salmonid restoration strategy (Hendry *et al.*, 2003). By targeting the processes that control habitat degradation such projects provide a more sustainable solution. For example, excluding livestock from the riparian zone can result in reduced bank erosion due to stock trampling, thereby reducing fine sediment delivery and hence siltation of gravels; encourage regeneration of bankside vegetation providing cover, shade and invertebrates for fish; and lead to improvements in channel morphology, habitat diversity and the accumulation of gravels (Opperman and Merenlender, 2004).

It is not just the riparian zone that is important. There is growing recognition that controlling processes can operate at the scale of the entire landscape (Wang *et al.*, 2003; Cowx and de Jong, 2004). This is particularly true for the case of water quality. Traditionally, fisheries management has focused on high-magnitude, point-source pollution located within the immediate vicinity of the symptoms such as sewage treatment outfalls and industrial effluent. However, progress has been made in reducing inputs from these sources which are now heavily regulated by statutory bodies such as the Environment Agency in the England and Wales. More attention is now being directed at diffuse pressures. This includes pollutants associated with land management and agricultural intensification, such as fertilisers (Heaney *et al.*, 2001), pesticides (Waring and Moore, 2004) and insecticides (Lower and Moore, 2003), microbial pathogens (Oliver *et al.*, 2005), and fine sediment (Soulsby *et al.*, 2001). There are also other diffuse pollutant sources such as salts, heavy metals, and hydrocarbons associated with road runoff (Barbee *et al.*, 2004; Sanzo and Hecnar, 2006), and acidification due to acid rain (Schindler, 1988). The impact of such pressures on in-stream water quality is controlled by catchment-scale processes such as land management, topography, and geology and climate. The location of pollutant sources within the landscape structure, flow pathways, and hydrological connection between sources and the channel network are also important in determining their impact (Burt and Pinay, 2005). Consider the issue of fine sediment delivery. Sources such as ploughed fields may be widespread in the landscape, but not all sources pose an equal threat to watercourses. Rather, risk is determined according to catchment-scale hydrological processes and the degree of connectivity exhibited between sources and the drainage network. The combination of sources and catchment-scale processes that contribute to the water quality at a particular point in the channel network are also important (Lane *et al.*, 2006). For example, an individual drain may contain a high level of suspended solids. If this is diluted by high quality water from several other inputs, or de-coupled from the channel by a buffer zone, there may not be a water quality problem at that site. Conversely, several inputs with moderate sediment levels that are highly connected to the channel and lack any buffering may combine to generate a fine sediment delivery problem at a particular site even though they were not initially identifiable as a problem themselves. Similar catchment-scale principals of landscape structure, flow path organisation and functional 'connectedness' can also be applied to case of solute transport within catchments (Burt and Pinay, 2005). In either case, critical sources of such pollution may only be identified with consideration of catchment-scale processes.

Despite the recognition that habitat controlling processes operate across a range of scales, it is often noted that few studies actually incorporate more than one scale of controlling process (Folt *et al.*, 1998; Wang *et al.*, 2003). Even where studies do incorporate more than one scale of process, they typically focus on trying to identify which scale is most important overall, which can lead to contrasting results. For example, Wang *et al.* (2003) suggest in-stream processes explain most variation in fish populations compared with catchment-scale and riparian-scale processes. Stauffer *et al.* (2000) conclude that riparian-scale processes are most important. Such conflicting results lead to confusion for fisheries managers who are trying to determine the most effective restoration strategy to adopt. Instead, the question fisheries managers need answering is not which scale of process explains most variation overall but which process is limiting where and, as Section 1.2.4 considers, over what spatial extent.

1.2.4 The need to consider the scale of investigation

The relationships identified between habitat and salmonid populations may depend upon the scale at which the investigation is undertaken. In this context scale refers to the spatial extent over which processes and relationships are measured (e.g. national, regional, catchment tributary, reach). Within this thesis this has been separated into three distinct scales (catchment-scale, area and tributary). A catchment-scale or catchment-wide approach relates habitat controls to variations in fish abundance data across the entire catchment. An area-scale approach focuses measurement upon capturing variability within a distinct area of the catchment (e.g. streams draining the Pennine escarpment) and a tributary approach considers relationships between habitat and variations in fish abundance within a single tributary. If salmon and trout abundance tracks a particular controlling process at one scale of investigation but not another, studies at different scales will arrive at different answers (i.e. there will be scale inconsistencies) (Folt *et al.*, 1998). As described by Stauffer *et al.* (2000), two studies in the same river basin by Lambert and Allan (1999) and Roth *et al.* (1996) found contrasting results. The first was undertaken in a small sub-catchment that had relatively similar land use throughout and found that in-stream habitat and local land use were most important in explaining ecological status. The second was undertaken over a much wider extent, which contained a wider variety of land uses, finding that regional land use was the most important factor. In this example, the larger the spatial extent of investigation, the greater the influence of catchment-scale processes on fish populations. This is not to say that processes operating at in-stream and riparian scales cannot exert significant influence over fish populations measured over a wide extent; rather it depends on the spatial extent over which these processes exert their influence. For example, an impassable barrier is an in-stream process but one which can influence a large extent of the catchment depending on its location within the river network. Similarly, whilst one isolated reach of stock access may not represent a significant pressure to populations catchment-wide, if stock access were to be found extensively throughout the catchment, it might accumulate to represent a major pressure.

Understanding the relationship of an organism with its environment requires understanding of the interaction between the heterogeneity of controlling processes observed within the environment and the scale at which an organism's response to that heterogeneity can be observed (Fahrig, 1992, *cited in* Folt *et al.*, 1998). It is particularly important to recognise this issue when undertaking research that is aimed at guiding management strategies. If the spatial scale of investigation does affect the degree of heterogeneity captured within different scales of habitat controlling process and, hence, does affect the habitat controls identified as limiting to salmonid populations, it poses an important dilemma for managers. It would mean that applying findings from research undertaken over one spatial extent to management undertaken at another could result in restoration that is ineffective and inappropriate. To take a hypothetical example, a catchment-wide research project may identify catchment-scale processes such as land management and diffuse pollution as the most important factor controlling salmonid populations. Conversely, research restricted in extent to an individual reach may find livestock access to be the most important factor. If the aim of restoration is to improve fish stocks catchment-wide, implementing bankside stock exclusion fencing catchment-wide may prove ineffective if the water quality remains poor as a consequence of diffuse pollution. However, if the aim of restoration is to improve populations within that individual reach (e.g. for the benefit of the fisheries owner), then addressing diffuse pollution may have little effect as water quality may not be limiting and addressing the issue of stock access may be of greater priority. Folt *et al.* (1998) suggest that "identifying inconsistencies in results via multiple scale observations is the first step for identifying the scales over which processes affecting distribution and abundance operate. Further, there can be many different organisations and individuals interested in undertaking fisheries habitat restoration within a river basin, each with their own very different aims. For example, angling clubs and riparian owners who may be particularly interested in fisheries performance within specific angling beats, regulatory organisations such as Natural England who may be most interested in improving habitat condition within designated areas (e.g. Sites of Special Scientific Interest), and Rivers Trusts who want to enhance biodiversity catchment-wide. It is therefore essential to understand the scale over which management aims are sought, before deciding on the scale of measurement and investigation required to inform that management.

1.2.5 Summary of research context

There are many and varied hypotheses for the perceived decline in salmonid fisheries. The hypothesis of habitat degradation has received particular attention, and a long history of habitat restoration and enhancement schemes has become established. To date, these have mainly been small-scale projects which, whilst achieving local improvements, have often failed to achieve the widespread catchment-wide increases in populations that are desired. This may be attributed to their failure to capture variations in the relationship between habitat and salmonids that occur through space and time. Four main scaling issues surrounding the relationship between habitat controls and salmonid populations have been identified. Figure 1.4 presents a conceptual representation of these issues, which are detailed below in terms of their implications for the minimum and maximum scale required by any investigation into the effects of habitat on salmonid populations. The numbers in Figure 1.4 correspond with the numbered points below.

1. Salmonids are mobile and make extensive use of the river environment. They move throughout the catchment and occupy different locations, at different stages in their life-cycle, as habitat requirements change. Consequently, successful and sustainable management may only be achieved with a catchment-wide approach to restoration that considers all these habitats. Without this approach, restoration may only succeed in redistributing populations throughout the catchment rather than increasing absolute populations catchment-wide.

2. The amount of habitat utilised by salmonids varies throughout their life-cycle as their level of mobility changes (e.g. 10s-100s metres for fry to 1000s kilometres for adults). This affects the spatial extent over which habitat variables impact salmonid populations. For example, at the minimum scale, fry are the least mobile life-stage and it is valid to make the assumption that their survival is strongly influenced by local conditions. Fry populations can therefore be related to habitat conditions at discrete spatial locations within the catchment. As mobility increases, survival is less dependent on local conditions and the extent over which habitat conditions must be evaluated increases. At the maximum scale, for smolt and adult survival, habitat conditions may need to be evaluated over many kilometres from their spawning site.

3. In-stream salmonid habitat conditions are controlled by a range of processes operating at different scales within the river basin (e.g. in-stream, riparian and catchment-scale). Processes operating at all scales have the potential to influence salmonid habitat at any point in the channel network and across any reach. Therefore, any catchment-wide study should include controlling processes from across all three scales..

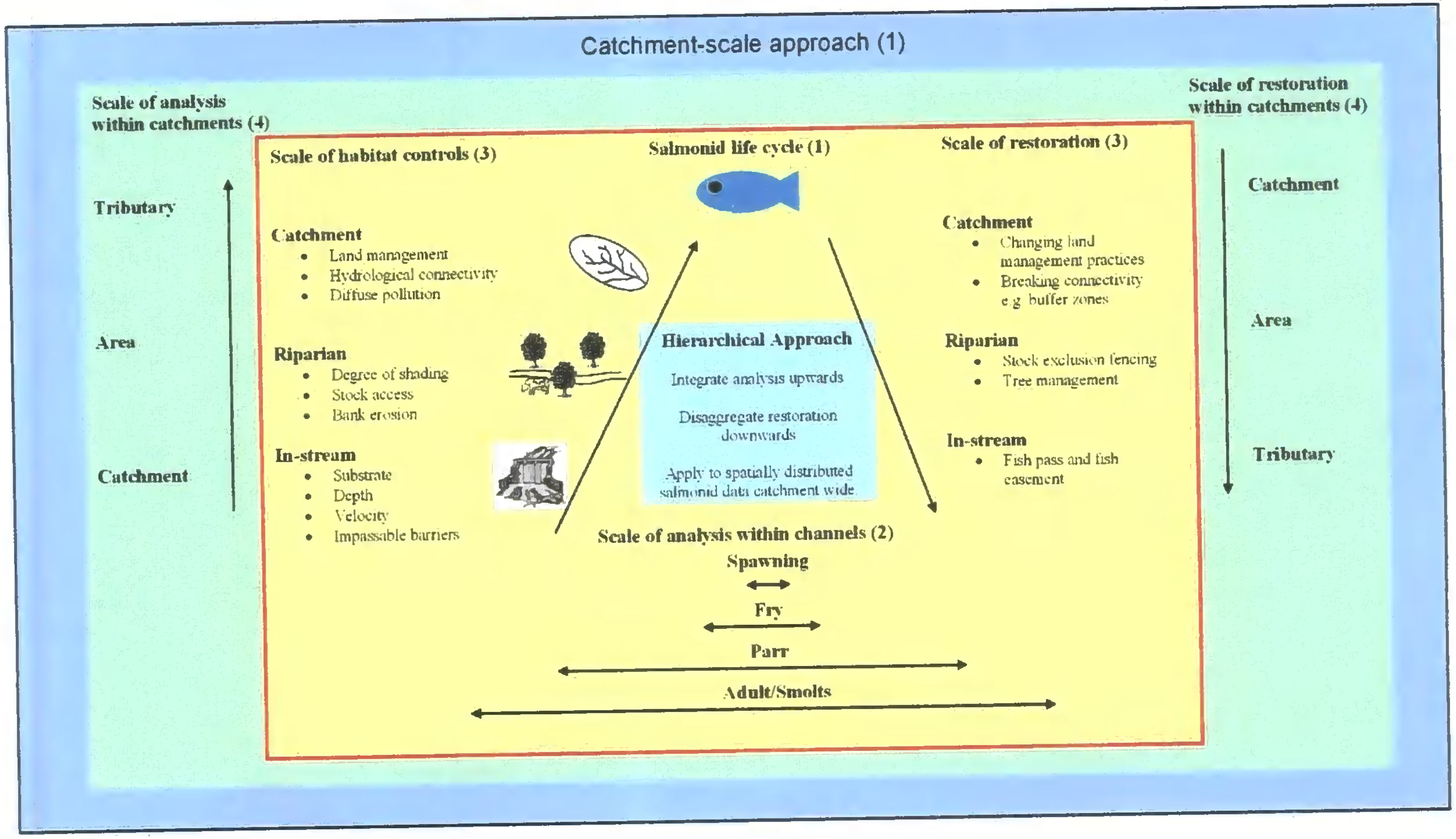

Figure 1.4: *Conceptual representation of the scaling issues surrounding studies on the controls of salmonid habitat.*

4. The importance of various habitat controlling processes may vary according to the spatial scale over which they are measured (e.g. tributary, sub-catchment, and catchment) due to different levels of heterogeneity within the environment at different scales of observation and in different locations. Within a catchment-wide analysis (as required by point 1), the minimum scale of investigation required for any life-stage is the catchment itself. This will yield broad-scale information on the major controls over salmonid habitat operating within the river basin. However, understanding variation in salmon and trout abundance at a finer scale, for example variations within individual tributaries as well as between tributaries, will require scientific investigation to be undertaken at a finer scale that is capable of capturing and measuring more subtle variations in habitat and salmonid response.. This typically requires a greater number of measurements to be made. For example, 40 sites may be required to capture the spatial variability in salmon fry populations within an individual tributary, and the same may be true for another 40 individual tributaries across the catchment. However, measuring spatial variability in salmon fry populations between those 40 tributaries may only require 200 samples rather than the 1600 cumulative samples required to capture spatial variability within all 40 tributaries (Williams and Hendry, 2003, p7).

The challenge facing researchers and fisheries practitioners is to determine how the interaction between habitat controls and salmonid populations can be studied, whilst taking into account these issues of scale, in order to identify the limiting controls and target habitat restoration more effectively. Within the scientific literature it has been proposed that this issue be addressed by adopting a hierarchical approach to research that transcends a variety of spatial scales, capturing intricacies, whilst not overlooking larger-scale processes (e.g. Wiley *et al.*, 1997; Armstrong *et al.*, 1998). However, a major obstacle to this type of research has, in the past, been the difficulty of acquiring spatially distributed data on salmonid habitat, sources of degradation and salmonid populations at the required scales. Given recent advances in remote sensing, GIS, environmental modelling, and in ecological surveying techniques such as electrofishing, there is now much more potential for overcoming this obstacle. Hence, the aim of this research is to couple recent advances in remote sensing, Geographical Information Systems (GIS), environmental modelling and ecological surveying techniques with current ecological understanding of habitat controls on salmonid populations, to help develop a more effective approach to prioritising habitat restoration.

1.3 Case Study – The Eden Catchment

Whilst this research is a generic study it is developed and applied to a case study of the River Eden catchment located in the North West of England (Figure 1.5). Adopting a case study approach provides a number of benefits (Shader-Frechette and McCoy, 1993). First, case studies facilitate holistic investigation of phenomena within their real life context, enabling evaluation of the practical control that may be achieved through habitat restoration. Second, ecological systems are typically unique, and precise deterministic rules developed through experimental and deductive science may not apply. Alternatively, case studies enable relationships, generalisations and hypotheses based on induction and statistical probabilities to be identified and developed that can subsequently be researched further, or used to aid in the description and investigation of other similar cases. Third, case studies allow concepts and theories to be tested in a specific location where the relevant variables cannot be controlled and manipulated as required by classical experimentation. As discussed, evaluation of entire catchments may be required. It is obviously impossible to control conditions over such a large-scale, yet some level of control can be achieved within a case study through the application of stratified sampling.

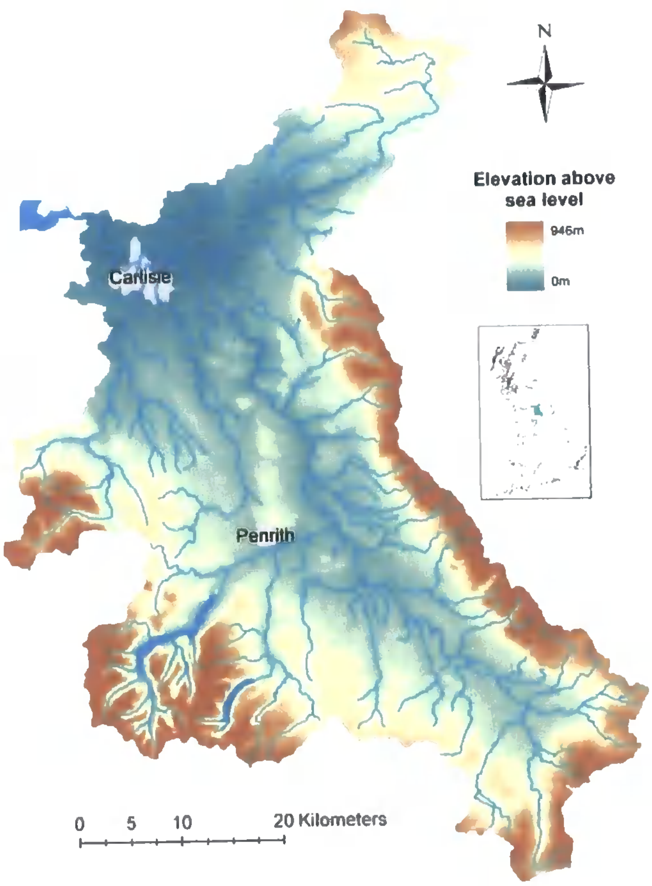

Figure 1.5: *The River Eden catchment. © Crown Copyright. All rights reserved 2006.*

Approximately 2,300km^2, the Eden catchment exhibits a diverse range of physical, ecological and topographic conditions: from sandstone to limestone geologies, oligotrophic to mesotrophic ecological conditions and high fell to lowland floodplain (Parsons *et al.*, 2001). These in turn support a diverse range of habitats and species, for example, 184 aquatic plant species have been reported on the Eden, more than any other river in the UK (JNCC, www.jncc.gov.uk). In addition to the landscapes, habitats and species, the Eden catchment and its water bodies are also a very important resource for the many people who depend upon it to provide water for domestic and industrial use, productive land for agriculture and attractions for tourism. Population density within the catchment is low and mainly concentrated in the city of Carlisle, together with the small towns of Penrith, Brampton, Appleby, and Kirkby Stephen, the remaining population being scattered throughout the catchment in numerous small villages and hamlets. Predominantly rural, 90% of the Eden catchment is under agricultural production. Consequently, issues relating to agriculture and its intensification are often cited as the cause of ecological degradation. However, agriculture within the Eden catchment is just as diverse as the physical landscape within which it is set and, therefore, so are the issues associated with it. A wide range of farming practices and types are found, from upland hill to lowland dairy, stocking (including, beef, dairy, sheep, poultry and pigs), to cropping (grass, cereals, and root crops). Tourism is the second major industry and accounts for 18% of Cumbria's economic output (Mackay Consultants, 2003). Therefore, maintaining the beauty of the catchment's landscapes, its ecological diversity and heritage is vital to the catchment's economy. The diversity of the Eden river basin makes it an excellent choice because it allows a wide range of habitat types, and spatial variability in habitat controls to be captured and investigated in terms of their impact upon salmonid populations. The catchment also comprises six distinct sub-catchments (herein referred to as the area-scale), varying from each other in terms of geology, topography, climate and land cover (Figures 1.6 (a-g). These areas can then be sub-divided further into individual tributaries. This facilitates the development of a hierarchical approach as investigation can be readily applied at the catchment, area and tributary-scale.

Figure 1.6(a): *Areas and tributaries of the Eden catchment*

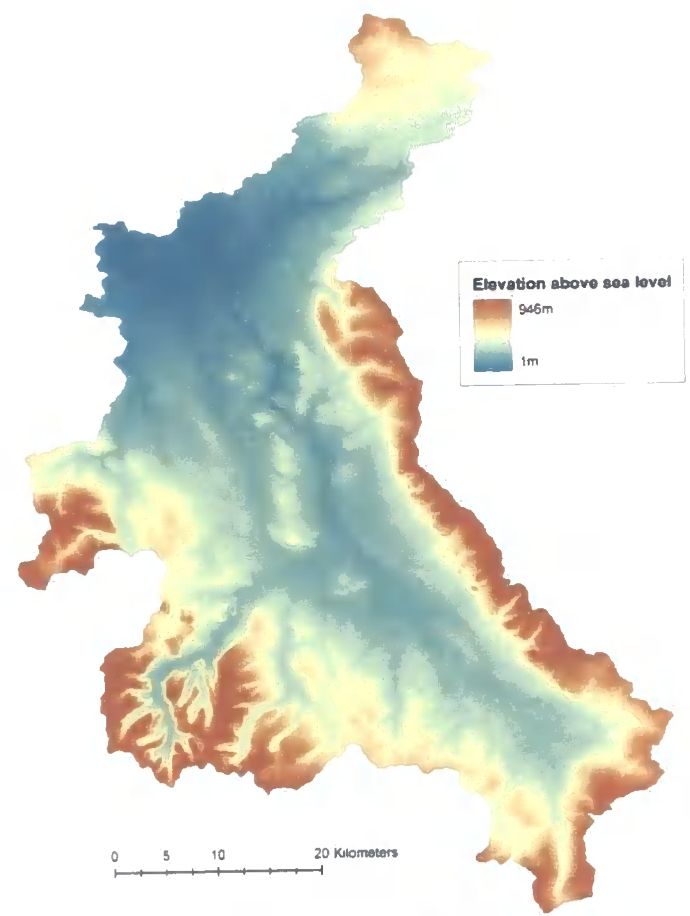

Figure 1.6(b) *Topography* (**Source:** *NEXTMap Britain ™ 5m resolution Digital Terrain Model*).

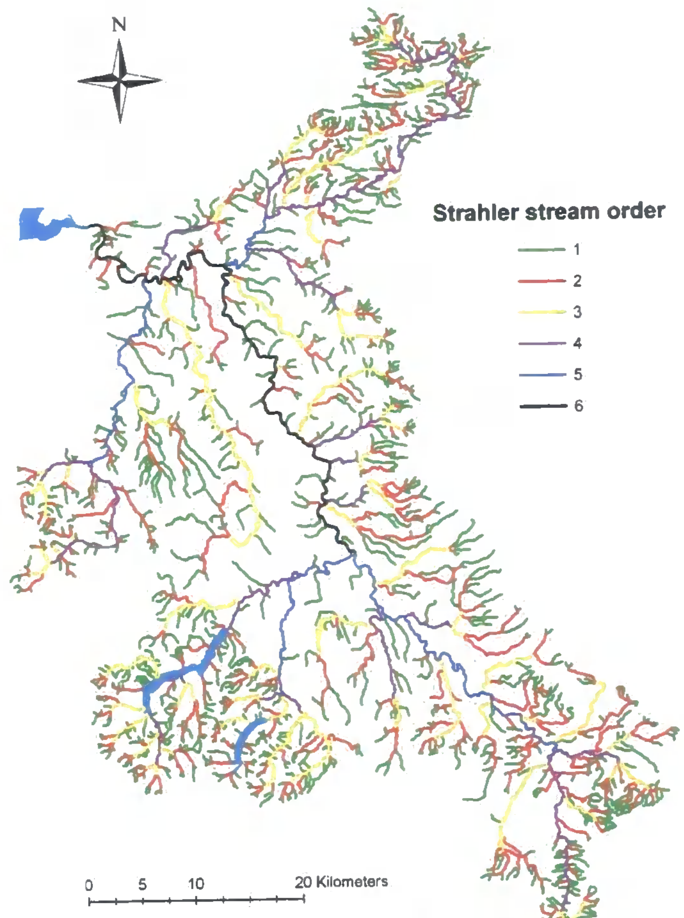

Figure 1.6(c) *Strahler Stream Order* (**Source:** *Digitised from OS Raster 1:50,000 Environment Agency, 100026380 (2007).*).

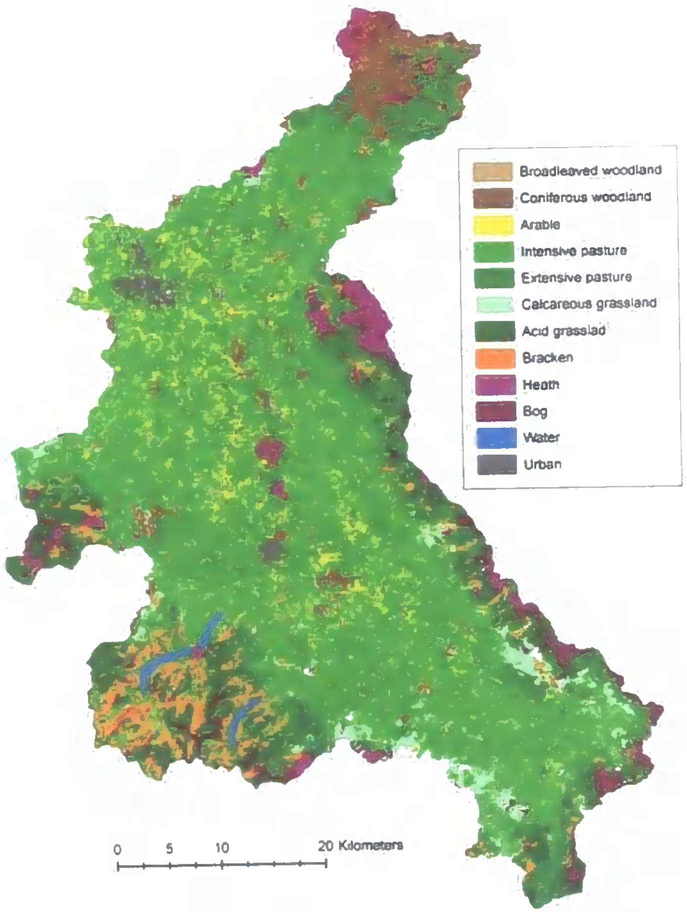

Figure 1.6(d): *Land cover (**Source:** Centre for Ecology and Hydrology Land Cover Map 2000)*

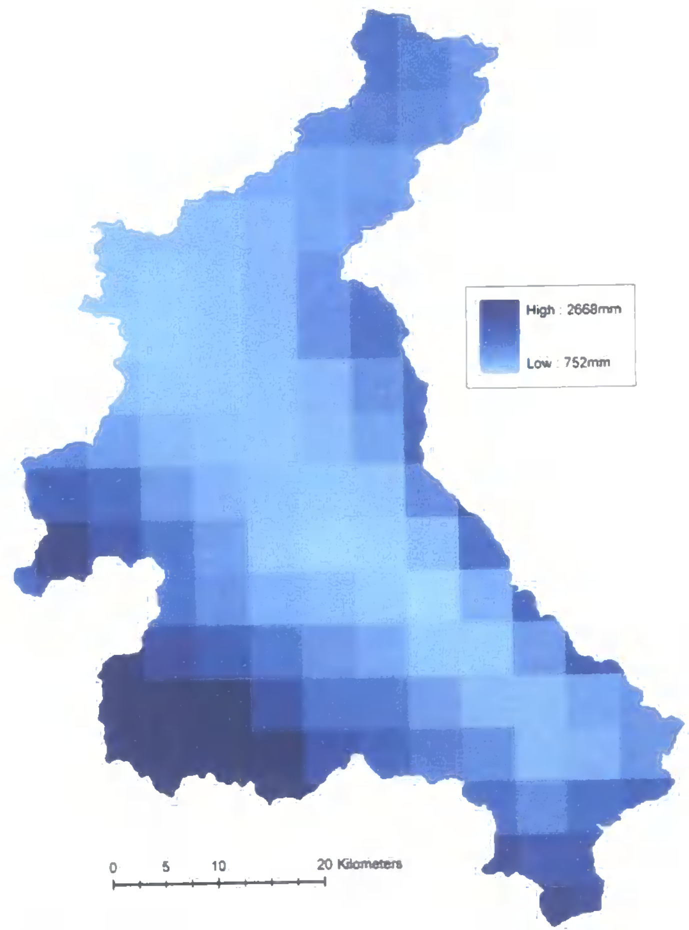

Figure 1.6(e) *Mean annual rainfall (**Source:** 1961-2000 baseline, 5km resolution, gridded dataset UK Meteorological Office).*

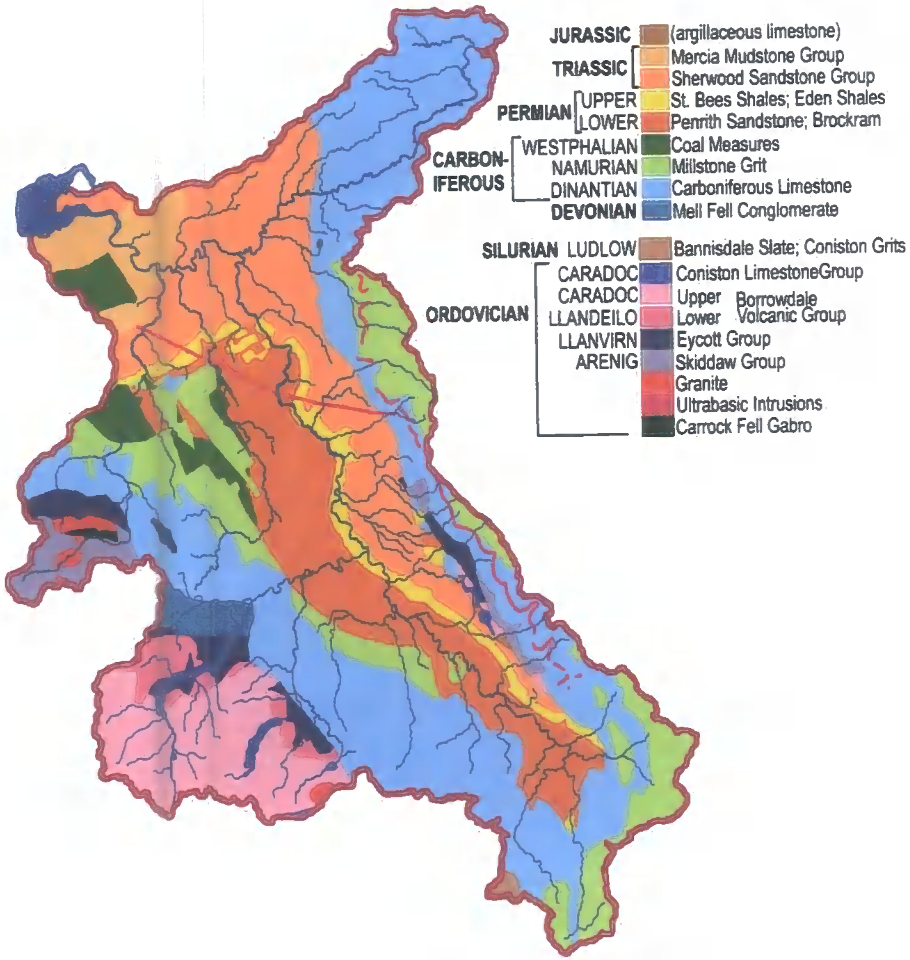

Figure 1.6(f): *Geology* (**Source:** *Eden Rivers Trust*)

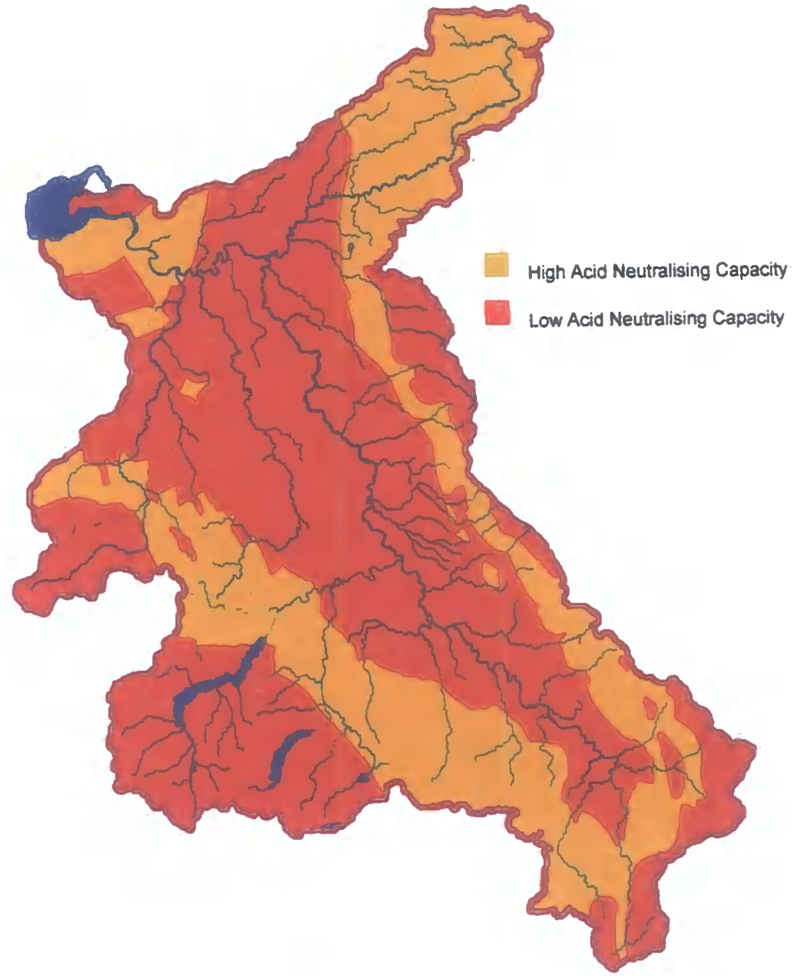

Figure 1.6(g): *Geological Acid Neutralising Capacity* (**Source:** *Classification based on Hornung et al. (1995))*

1.3.1 Detailed site description

Each of the distinct sub-catchments (areas) of the Eden catchment is described below.

Upper River Eden: In the south of the catchment, the area upstream of Stenkrith Falls (impassable by salmon and trout) near Kirby Stephen is termed the 'Upper River Eden' area for the purposes of this project. Here the River Eden, a dynamic gravel-bed river, meanders through the Mallerstang valley where it is confined to a relatively narrow floodplain bounded by the steep fells of Mallerstang Common (>550m above sea level). Draining extensive areas of blanket peat are numerous, fast flowing upland tributaries characterised by cascades, waterfalls and oligotrophic conditions. The dominant land use is extensive hill sheep farming predominantly on common land. Further downstream towards Kirby Stephen, the valley floodplain opens out as the River Eden flows over first limestone and then sandstone geology. Here, more intensive stock rearing of both cattle and sheep on permanent pasture and loam soils is found

Orton & Howgill Fells: Also in the south of the catchment, rising on Carboniferous Limestone, the Scandal Beck, Helm Beck, Hoff Beck, River Lyvennet and River Leith drain the Orton and Howgill Fells. These tributaries are typified by high levels of dissolved calcium carbonate with characteristics more akin to the Chalk streams of southern England (Coleman, 2003). Rich in nutrients, they exhibit mesotrophic conditions and are particularly important for the endangered white-clawed crayfish which requires calcium carbonate for its carapace. The topography of this area has a tendency towards wide, gently sloping valleys, and this together with the nutrient-rich soils, has resulted in an area that is intensively farmed. This includes dairy, and in more recent years an expansion in cropping including winter cereals and maize has been seen.

Pennine Becks: To the east of the catchment, the Northern Pennines are capped by the Great Whin Sill, an igneous rock, together with limestone in the uplands and soft red sandstones in the lowlands. The topography and aspect of the steep Pennine escarpment, 893m at Cross Fell, makes this area prone to high rainfall. The streams draining this area are highly dynamic, sinuous, fast flowing upland becks with a plentiful supply of coarse sediment. They represent some of the most ecologically productive streams within the Eden catchment. The uplands are typified by extensive peat moorland and blanket bogs, and extensive hill sheep farming is the dominant land use. This gives way to more intensive stocking of both sheep and cattle together with some cropping in the lowlands were the climate is less harsh and soils are more free draining (e.g. sandy loams).

Ullswater and Lowther Valley: In the south west of the catchment older, more resistant, volcanic rocks of low acid neutralising capacity form the high Lake District fells. The geology of this area leads to nutrient-poor soils, acidic becks and oligotrophic conditions. The topography results in steep upland becks that are fast flowing and characterised by waterfalls, cascades and rapids, many of which drain into Ullswater, the second largest still water in the Lake District. This is then subsequently drained by the River Eamont, one of the major tributaries of the river Eden. This area experiences the highest rainfall within the catchment, with the long-term average (1941-1970) recording 3,031mm p.a. at the Blea Water rain gauge (OS ref: NY456108). Within this marginal environment agriculture is predominantly extensive with enclosed more intensive farmland confined to the valley floor. The River Lowther is also included in this area. It is heavily regulated with two large impoundments, the Haweswater and Wet Sleddale reservoirs in its headwaters. As such, it experiences an attenuated flow regime. This area is renowned as the predominant spawning ground for multi-sea-winter, spring-run salmon (Gowans, 2004).

The Tyne Gap: In the north of the catchment the highly dynamic and sinuous River Irthing and its tributaries drain areas of extensive peat moorland, with acidic soils. These rivers flow through deeply incised gorges within their middle reaches and onto sandstone floodplains before their confluence with the River Eden just east of Carlisle. Coniferous forestry is the dominant land use in the uplands of the Irthing catchment and this combined with the peat moorland leads to acidic streams that are high in dissolved organic carbon and highly coloured. The lowland floodplains are more intensively farmed and dominated by beef cattle. Relatively undisturbed stands of alluvial forest (Alder and Willow) are associated with the Irthing catchment where they occur on the shingle and gravel of actively migrating channels.

Caldew and Petteril Rivers: To the west of the catchment two major tributaries of the River Eden, the River Caldew and River Petteril both flow into the Eden at Carlisle. The River Caldew drains the Skiddaw fells exhibiting characteristics similar to the Ullswater tributaries in its headwaters. Through its middle reaches, the Caldew valley and its tributaries the River Roe and River Ive, are subject to some of the most intensive farming, primarily dairy, within the Eden catchment, whilst the lower reaches are heavily modified reflecting their industrial, past and present, and the need for flood protection. The River Petteril is a lowland meandering river with gentle riffle-pool sequences throughout its length. Similar to the River Caldew it is subject to intensive farming through its middle reaches with lower reaches that are heavily modified. The River Petteril has the added pressure of contaminated runoff from the M6 and A6 which run alongside it for much of its length.

Ultimately, these six distinct sub-catchments all drain into the main River Eden, characterised predominantly by sandstone geology, and nutrient-rich waters. The lowland valley landscape is varied, consisting of areas of extensive open floodplain where fertile loam soils are intensively farmed by a combination of dairy, stocking and cropping, to deeply incised sandstone gorges with broadleaved woodland.

1.3.2 Institutional management framework

The diversity of the Eden catchment has led to much of the landscape, its heritage, river and tributaries being designated and protected under national and international law (Figure 1.7 and Table 1.2). However, within the UK, there is no one organisation responsible for managing all aspects of the environment in general or catchments in particular. This has led to the development of a complex institutional framework from central government right down to individual communities and land managers (Table 1.3).

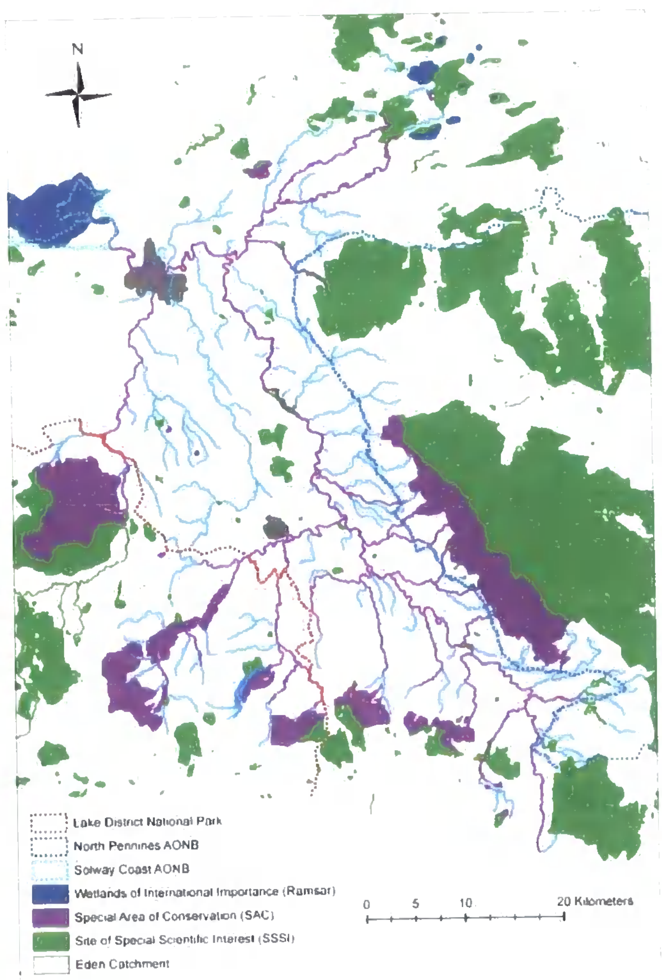

Figure 1.7: *Designations under national and international law applying to the River Eden catchment. © Crown Copyright. All rights reserved 2006.*

Table 1.2: *National and international laws applying to the environment of the River Eden catchment*

Laws & Designations	Description
National Designations	
Wildlife and Countryside Act, 1981 Site of Special Scientific Interest (SSSI)	The River Eden and tributaries is noted for floating vegetation of plain and sub-mountainous rivers (*Ranunculus* spp.), white-clawed crayfish, sea, brook and river lamprey, bullhead, otter and the Atlantic salmon
The National Parks and Access to the Countryside Act 1949	Under this act the Lake District National Park, North Pennines Area of Outstanding Natural Beauty (AONB), Solway Coast AONB, and a number of National Nature Reserves and Local Nature Reserves have been established
European Designations	
Directive 92/43/EEC, Conservation of Natural Habitats and of Wild Fauna and Flora. Special Area of Conservation (SAC)	Extends the level of protection provided under the SSSI notification to include residual alluvial woodland and cover an area of 2,550 ha.
Directive 2000/60/EC, Water Framework Directive	Under the Directive all European waters must achieve 'good ecological status' by 2015
Directive 79/409/EEC, Conservation of Wild Birds	In the north east of the catchment is the Geltsdale Reserve, managed by the Royal Society for the Protection of Birds (RSPB)
International Designations	
The Ramsar Convention 1973 'Wetlands of International Importance'	Rockcliffe Marshes, areas of the Solway Firth, Butterburn Flow and Kielder Mires
Convention Concerning the Protection of the World Cultural and Natural Heritage, UNESCO 1972 'World Heritage Site'	Hadrian's Wall. There are also numerous other Scheduled Ancient Monuments within the catchment, many of which are believed to have strong connections with water and represent the historic importance of water within the landscape

Table 1.3: *Institutional management framework within the Eden catchment*

Institution	Examples	Description
Central government	Department for the Environment, Food and Rural Affairs (DEFRA)	
Statutory public bodies	Environment Agency (EA) Natural England (NE)	Amongst other responsibilities the EA are the competent authority[1] in England and Wales for delivering the Water Framework Directive. NE are responsible for maintaining SSSIs and SACs in favourable condition
Non-departmental public bodies (NDPBs)	Lake District National Park Authority and the North Pennines and Solway Coast AONB partnerships.	Manage and co-ordinate conservation efforts within designated and protected areas of the catchment. They rely on central government and the statutory public bodies for funding.
Local government authorities	Cumbria Country Council; Eden District Council; Carlisle City Council	Responsible for planning, refuse collection, and implementing national planning policy guidance.
Non-governmental organisations (NGOs)	Eden Rivers Trust; RSPB; Cumbria Wildlife Trust; Flora of the Fells; Friends of the Lake District; National Trust	Cover a diverse range of environmental and conservation remits
Private water companies	United Utilities	Responsible for the provision of safe, clean water for domestic and industrial use and for the cleaning of waste waters after use.
Individuals	Landowners, farmers, anglers, businesses and communities	All may have an interest in ensuring sustainable use of the catchment's resources.

[1] The Eden catchment falls within the Trans-national boundary between England and Scotland where the Scottish Environmental Protection Agency (SEPA) are the lead competent authority responsible for delivering the Water Framework Directive.

All the organisations and individuals included in Table 1.3 cover different geographical areas and have different responsibilities, management aims and objectives resulting in catchment management that can be disparate and inefficient, and that frequently leads to conflicts between managers. Whilst each institution may be achieving benefits in its own area of responsibility, or geographic location, its actions may be damaging to another or simply missing opportunities to deliver mutual benefits. A classic example of conflicting management is the use of hard engineering such as revetment, canalisation and barrages for flood defence, all of which are well documented as being damaging to ecology and biodiversity. Instead what is required, as enshrined in the Water Framework Directive, is holistic management of water and people within the river basin (catchment) framework.

One group of Non-governmental organisations (NGOs) within England and Wales that is striving to adopt a catchment approach to river restoration is the Rivers Trust movement, which includes the Eden Rivers Trust. Covering entire river catchments, Rivers Trusts are ideally placed to achieve this goal as they are not limited to specific geographical locations within catchments. They also have no statutory powers, legal authority or responsibilities. Instead, they work on the basis of voluntary partnerships with many organisations and as such are better placed to secure the agreement to, and involvement of, local communities and individuals in restoration projects. The national Association of Rivers Trusts (ART) says: "Rivers trusts have been described as having "wet feet" because they have the reputation of being "doers" concentrating much of their effort on practical catchment, river and fishery improvement works on the ground" (www.associationofriverstrusts.org.uk).

1.3.3 The Eden Rivers Trust

The Eden Rivers Trust (ERT) is a charitable organisation (charity number 1059534) formed in 1996 and has the following mission statement:

"To secure the conservation, protection, rehabilitation, and improvement of the rivers, streams, watercourses and water impoundments, together with their related banksides and estuary with respect to the River Eden (Cumbria) its tributaries and the Eden Valley. To advance the education of the public in the management of water and water habitats."

The principal aims of ERT are:

- To secure the conservation, protection and rehabilitation of the River Eden Catchment; and
- To promote education and understanding of the river environment.

In terms of delivering these aims, ERT is currently involved with its partners in a catchment-scale restoration plan 'Restoring Eden'. The Trust recognises that with limited funds it is prohibitively expensive and impractical to restore every habitat within the catchment. Instead, ERT has adopted a catchment-wide, ecosystem approach, aimed at delivering conservation, protection and rehabilitation of the River Eden catchment that is targeted to achieve the greatest cost-benefits. Simultaneously, the Trust's education programme aims to promote awareness, increase understanding and foster stewardship of the river environment to encourage and ensure the sustainability of restoration projects throughout future generations. Restoration targeting is based upon sound scientific research and best practice principles, which to date have involved:

1. A socio-economic impact assessment into the current value of the River Eden and likely impact of restoration to the local economy (Mackay Consultants, 2003);

2. Annual catchment-wide semi-quantitative electrofishing surveys undertaken since 2002. These are targeted at salmonid fry habitat, with the aim of identifying reaches of the River Eden and its tributaries where environmental degradation is occurring (Section 3.3.1); and

3. Quantitative electrofishing surveys in 2005 aimed at collecting more detailed data on salmonid populations, including parr, within specific habitats, and monitoring the impact of restoration projects on aquatic ecology (Section 3.3.2).

Through collaboration with the Eden Rivers Trust, this research brings together the Trust's knowledge of salmonid population distribution and abundance with data on the spatial distribution of salmonid habitat controls, the mutual aim being to develop a catchment approach to targeting fisheries habitat restoration that is practical and efficient. Local knowledge of the Eden catchment, its fishery and land management issues has been greatly enhanced by working with the Trust.

1.3.4 Salmonids in the Eden catchment

The River Eden supports an important and diverse fishery, estimated to contribute £1.2 million to the local economy per annum through angling alone (Mackay Consultants, 2003), of which the salmonid species of Atlantic salmon and brown trout are an important part. Once renowned as the best salmon fishery in the whole of England, the Eden boasts the record for the largest salmon ever caught on an English river (Figure 1.8).

Figure 1.8: *The largest rod caught salmon in England: Caught by Mr Lowther Bridges in 1888, the fish measured 54" in length, had a girth of 27" and was estimated to weigh 56lbs (pers comm. R. Coleman).*

However, it is now the general opinion, primarily based on anecdotal evidence that stocks of both Atlantic salmon and brown trout within the catchment are in decline. Below is a selection of typical anecdotes from local anglers who have fished the River Eden over the last 50 years.

"...there has been a general decline in sport on all of our local rivers over the years. Who says so? Every experienced angler of my acquaintance who remembers the fishing of the 1950s, 1960s and 1970s"

"The river Petteril, our earliest trout river, which used to produce trout in table condition well ahead of its larger sister tributaries of the Eden, is now written off as a trout fishery"

Terry Cousin, Fly-fishing and Fly-tying, March 1998.

"I have represented the interests of my local club, Penrith Angling Association as a committee member for over 30 years and have observed a steady decline in the trout population"

"Between the 1960s and the 1990s the effects of all the above factors[2] have contributed to a consistent downward trend in the general ecology of the Eden catchment area and the fish populations in particular. There are many streams now which used to hold high numbers of fish where today one will be very fortunate to find any. ..The fly life has been reduced to a fraction of what it was and flies such as Mayfly, March brown, Iron blue, Grannom, Creeper, some Sedges etc are hardly ever observed on the river"

J.S. Kinnear, Letter to the Right Honourable David Maclean, House of Commons, December 1997.

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[2] The factors listed in Mr Kinnear's letter included UDN Salmon disease in 1965/1966; land drainage; tree removal; canalisation; pollution events; land use intensification; and predation.

Quantitative data regarding historic salmonid stocks in the Eden catchment are much sparser. Whilst there have been a few isolated studies undertaken on small tributaries within the catchment (e.g. Crisp and Cubby, 1978), data at the catchment-scale and over a continuous period of time are lacking. Data on salmon rod catches in the Eden have been collected by the Environment Agency since 1990, as declared on rod licence returns (Figure 1.9) and these have been used, together with information from a fish trap on the River Caldew, to estimate salmon egg deposition over the last 15 years (Figure 1.10). Egg deposition would appear to have dropped considerably during the period 1997-2003, failing to reach management targets, but higher levels were once again seen in 2004. Just how these rates relate to salmon stocks over longer periods of time is unclear.

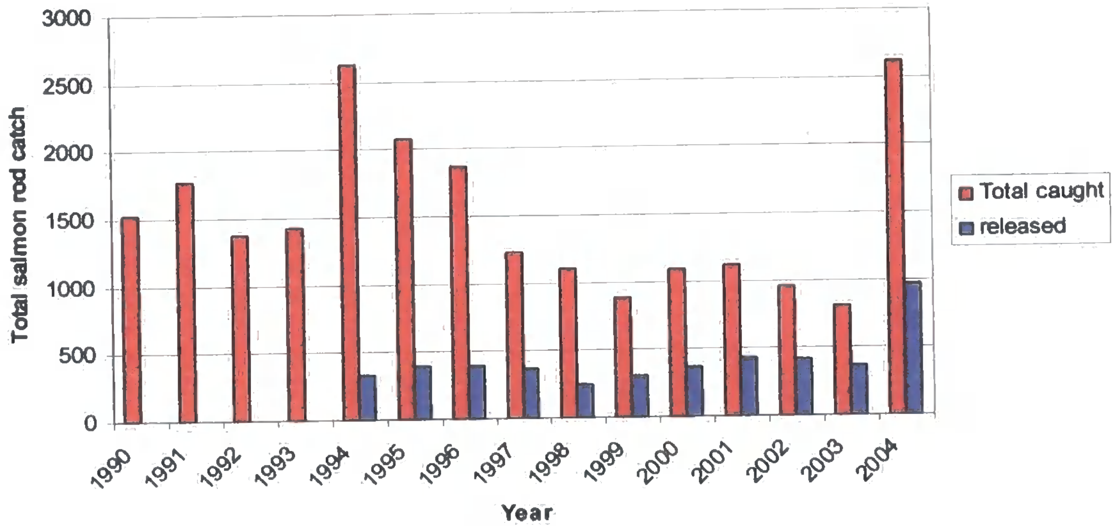


Figure 1.9 *Atlantic salmon rod catch as declared on Environment Agency rod licence returns. (Courtesy of the Environment Agency).*

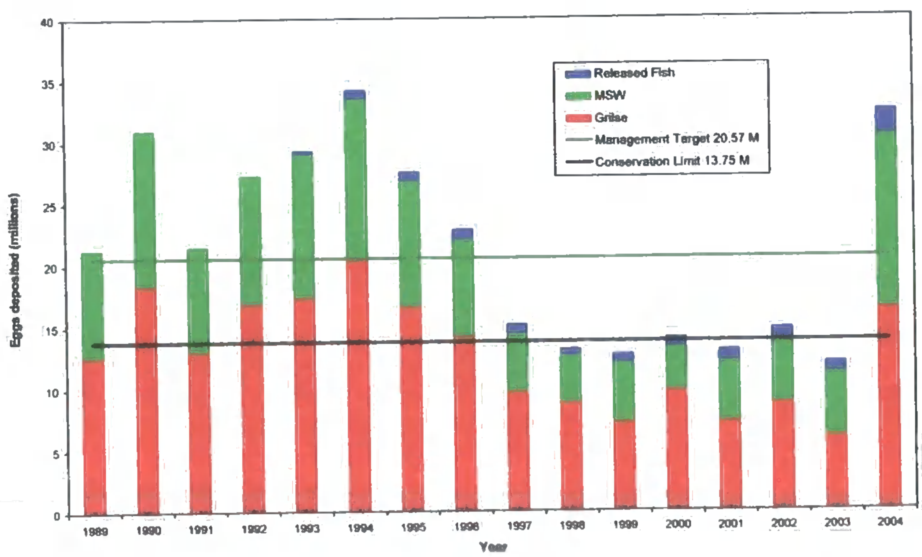


Figure 1.10: *Estimated Atlantic salmon egg deposition in the Eden catchment. (Courtesy of the Environment Agency).*

The level of data on current stocks in the Eden is much higher, and represents one of the most detailed spatial datasets on stocks at the catchment-scale in the UK. This comprises information from catchment-wide semi-quantitative electrofishing surveys undertaken by the Eden Rivers Trust since 2002, complementing the Environment Agency's own electrofishing programme; data from the Environment Agency's resistivity fish counter located at Corby on Eden (OS Reference: NY 469545); and data on spawning locations collected via the Environment Agency's radio-tagging programme (Gowans, 2004). Further details the collection of fisheries data in the Eden catchment are presented in Chapter Three.

1.3.5 Summary of the case study approach

Based on a case study of the River Eden catchment, this research has been undertaken in collaboration with the Eden Rivers Trust, its aim being to couple recent advances in remote sensing, Geographical Information Systems (GIS), environmental modelling and ecological surveying techniques with current ecological understanding of habitat controls on salmonid populations, to develop a more effective approach to prioritising habitat restoration. The scope of the research has been set by the fisheries data available. As such, this research will adopt a catchment-wide approach focused primarily on the relationship between habitat controls and the salmonid species, Atlantic salmon and brown trout at the fry life-stage. Analysis will be undertaken initially at the catchment-scale. The Eden catchment can be divided into 6 distinctive areas based on geology, topography and land use. Investigation of whether the relationships identified between habitat controls and salmonid populations vary between these areas will also be undertaken, and where data permits, analysis will be extended to the tributary-scale. Research will also be conducted into the relationship between habitat controls and salmonid parr for an area of the upper catchment.

1.4 Detailed research objectives and thesis structure

The aim of this research is to be achieved through three main objectives, discussed in detail below.

1.4.1 Objective (1): *To review and to synthesise current understanding of in-stream, riparian and catchment-scale controls on freshwater salmonid habitat throughout the life-cycle of salmon and trout, to help formulate a set of hypotheses for further investigation.*

The ultimate aim of this research is to develop a more effective approach to informing and prioritising salmonid habitat restoration at the catchment-scale. This is to be achieved by investigating the relationship between salmonid abundance and a range of habitat controls at the in-stream, riparian and catchment-scale. Consequently, a fundamental factor constraining the success of this research is the ability to identify and select, from the outset, a suite of potential habitat controls that are considered to be ecologically relevant in terms of salmonid performance, and about which habitat restoration decisions are currently, or can be made. To achieve this, a review of the current scientific literature is required to examine expert opinion and current understanding about which controls at the in-stream, riparian and catchment-scale may exert an influence over salmonid performance and the mechanisms by which they exert that influence. However, the sources of research surrounding this issue are diverse and cover a number of scientific disciplines.

Ecological research in general has made significant advances in understanding the mechanisms by which natural populations are regulated. The theories of stock recruitment, carrying capacity, density-dependent and density-independent mortality and self-thinning are well documented and form the general consensus within the scientific community. These theories have been applied to studies of salmonid population dynamics and a comprehensive review of current understanding in this area was presented by Milner *et al.* (2003) at the Salmonid 21C conference. However, there is some debate on the issue of population regulation and density dependence in particular (Berryman *et al.*, 2002). A number of researchers (e.g. White, 2001) oppose these theories and their alternative hypotheses should be explored.

Fisheries scientists have made considerable advances in understanding the in-stream habitat requirements of salmonids (e.g. see Armstrong *et al.*, 2003 for an in-depth review), in terms of depth, velocity, substrate and cover at different stages throughout their life-cycle and a summary of this was presented in Table 1.1. Substantial research into the water quality requirements of salmonids has also been done (e.g. Heaney *et al.*, 2001; Waring and Moore, 2004; Lower and Moore, 2003). However, much of this work has been undertaken through laboratory experimentation where habitat properties and water quality can been artificially controlled. Whilst this research provides essential information about the environmental in-stream conditions that are important to salmonid survival what is less well understood within ecology are the controls over these conditions and the mechanisms by which they operate within the natural environment.

At the opposite end of the spectrum research concerning the environmental controls of in-stream morphology, flow and water quality has been predominantly the realm of fluvial geomorphologists and hydrologists. Again, significant progress has been achieved in these areas. The theories of connectivity and coupling between the riparian zone, wider catchment and channel network, and mechanisms by which energy, water, sediment and pollutants are transferred throughout the environment are well documented. Beginning in the 1970s with Schumm's (1977) work on fluvial geomorphology and hillslope channel coupling, and Brunsden and Thornes' (1979) work on landscape sensitivity, these theories have since been adapted, applied and developed to work on sediment delivery (e.g. Walling, 1983), buffer zones (e.g. Haycock and Burt, 1993,) the flood pulse concept (e.g. Middleton, 1999), and the delivery of material from hillslope failures (e.g. Harvey, 2002). More recently researchers have raised the theory of hydrological connectivity at the catchment-scale as a tool for determining the impact of certain land management strategies upon the delivery of water, sediment and pollutants to the channel (e.g. Burt, 2001; Lane *et al.*, 2003a) and as a concept for integrating dynamic runoff generation with landscape characteristics to investigate and model flood production (Bracken and Croke, in press). The impact of certain land management practices upon these processes has also been the focus of much research, for example, the impact of animal stocking densities upon catchment hydrology, sediment generation, flooding and bank erosion (e.g. Trimble and Mendel, 1995; APEM, 1998, and Greenwood *et al.*, 1998). What is less well understood within these fields is which of these processes are the most ecologically significant in terms of fish.

As Summers *et al.* (1996) note, there is a lot of relevant but disparate knowledge within these disciplines which needs to be drawn together if sustainable restoration techniques are to be developed. Without these links it is impossible to identify which controls potentially have the greatest ecological significance or to determine what effect restoration strategies may have upon fish stocks. For example, siltation of spawning gravels is known to significantly reduce salmonid egg survival within redds, whilst catchment hydrological processes coupled with land management practices are known to influence fine sediment delivery rates. However, failure to make the direct connection between sources of habitat degradation such fine sediment delivery from agricultural fields and ecology has resulted in managers trying to treat the symptoms of such problems using short-term strategies such as gravel cleaning rather than implementing long-term sustainable solutions such as soil conservation strategies.

A mechanism for integrating and synthesising a large volume of data and research from across disciplines is therefore required. One such approach, that is being increasingly used by researchers and environmental policy makers such as the European Environmental Agency is the DPSIR (Driver – Pressure – State – Impact - Response) framework (e.g. Gobin *et al.*, 2004; Borja *et al.*, 2006; Karageorgis *et al.*, 2006). This approach provides a framework for identifying different factors (Drivers and Pressures) which may impact on salmonid habitat (State) and ultimately salmonid abundance and distribution at different stages of the life-cycle (Impact). Restoration strategies (Responses) may then be linked back to previous stages of the framework to consider their sustainability and potential impact upon salmonid populations. Typically used for integrating science with socio-economic policies the approach can also be adapted to integrate different branches of science to provide a simplistic but holistic conceptualisation of the system.

The focus of this objective therefore, is to review the scientific literature and apply the DPSIR approach to the Eden catchment to identify a suite of ecologically relevant habitat controls at the in-stream, riparian and catchment-scale, together with hypotheses about how those controls may impact salmonid populations for further investigation under Objective (3).

1.4.2 ***Objective (2):*** *To employ recent advances in remote sensing, GIS and environmental modelling, to identify, to develop and to validate tools for quantifying the habitat of salmon and trout at the catchment-scale, appropriate to each habitat control and scale of control.*

A fundamental data requirement to achieving the aims of this research is a catchment-wide survey of habitat controls across a range of scales. This requirement has previously been a major obstacle to undertaking this type of research due to the considerable costs and impracticalities involved in collecting data. However, with recent advances in remote sensing, GIS, and environmental modelling there is now the potential to overcome this obstacle. To this end, Chapter Three discusses the broad-scale data needs of this research, presenting the catchment-wide data sources that are available for the Eden catchment, including digital topographic data, satellite imagery and aerial photography. Chapters Four and Five then focus on identifying, developing and validating tools by which this data can be used to derive salmonid habitat information relevant to each habitat control and scale of control.

Chapter Four focuses on the derivation of in-stream and riparian scale habitat information. In-stream and riparian physical habitat is a product of the geomorphological processes of water and sediment transfer acting within the catchment (Newson & Newson, 2000). As such, most river habitat survey techniques focus on recording geomorphic forms and processes, and subsequently interpret these in terms of habitat availability for the species in question. Traditional fluvial geomorphological studies take a field-based approach (Sear, *et al* 2003); and numerous walkover survey protocols have been developed over the years, (Table 1.4) ranging from generic baseline surveys such as the Environment Agency's River Habitat Survey to user specific surveys such as APEM's walkover survey of salmonid habitats (Hendry and Cragg-Hine, 1997).

Table 1.4: *Traditional geomorphological survey techniques*

Survey	**Description**	**References**
River Habitat Survey (RHS)	Developed by the Environment Agency this semi-quantitative technique aims to characterise and assess, in broad terms, the physical structure of freshwater streams and rivers to provide a catchment baseline survey.	Raven *et al.* (1997) Parsons *et al.* (2001) Sear *et al.* (2003)
GeoRHS	An extension of the classic RHS, which facilitates the collection of more quantitative information on channel morphology and which is extended to the wider floodplain.	Sear *et al.* (2003)
Fluvial Audit	Overview of the sediment system typically aimed at addressing sediment-related management issues, and identifying sediment source, transfer and storage reaches within the river network.	Sear *et al.* (2003)
Geomorphological Dynamic Assessment	Field survey of channel form and flows; hydrological and hydraulic data, bank materials and bed sediments. Aimed at understanding reach dynamics and channel morphology.	Sear *et al.* (2003)
User specific surveys	Unlike the above generic surveys, several researchers and organisations have developed more specific surveys focused on recording only those features relevant to the survey's purpose, e.g. APEM's walkover survey for salmonid habitats.	Hendry and Cragg-Hine, (1997)

Whilst providing valuable information, at the catchment-scale and particularly for large catchments in excess of 1000 km^2, walkover surveys can become prohibitively time consuming and costly. There is therefore a real need to develop an alternative approach capable of delivering the information required in a more cost-effective manner. Table 1.5 considers some of the features typically recorded by walkover surveys and identifies alternative approaches currently documented in the scientific literature which are based on remote sensing and GIS. Techniques such as the interpretation of channel planform change from aerial photography (Winterbottom, 2000; Micheli & Kirchner, 2002) are now commonplace. However, the study of finer-scale features still remains in the realm of scientific research or if applied, is only done so for small catchments or small areas of catchments, primarily due to cost. As with all technology, remote sensing is undergoing rapid development with new sensors and datasets emerging all the time. Therefore, the aim of Chapter Four is to evaluate the capability of two recently developed datasets, 20cm digital aerial photography and the Nextmap Britain 5m digital terrain model (DTM), for undertaking river habitat survey at the catchment-scale.

Table 1.5: *Summary of features typically recorded during walkover surveys together with alternative approaches for collecting the same data using GIS and remote sensing.*

Feature	**Information recorded by ground surveys**	**Remote sensing and GIS studies**	**Location**	**References**	**Spectral Resolution**	**Spatial Resolution**
Channel planform	Sinuosity Channel dimensions e.g. width Channel migration	Change detection of width, braiding index and sinuosity from 1755 to 1994. Rates of channel migration mapped over a 40 year period.	Rivers Tay & Tummel, Scotland Kern River, USA	Winterbottom (2000) Micheli & Kirchner (2002)	OS maps; Colour aerial photography Colour aerial photography	1:5 000 to 1:36 000 1:10 000 to 1:20 000
Riparian vegetation and tree cover	Land cover within 50m & 5m of the channel Aquatic vegetation e.g. macrophytes Overshading Tree density e.g. continuous, scattered Woody debris	Classification of riparian vegetation type into 7 classes including conifer, hardwood, brush and pasture. Automated mapping of woody debris	Yaquina River, USA Lamar River USA	Congalton *et al.* (2002) Marcus *et al.*, (2003)	Colour aerial photography 128-band imagery	 1m pixels
Bank erosion	Type & failure mechanism Bank profile e.g. undercut Bank structure and material	Bank erosion rates (from aerial photography) were related to field data on bank profile, structure and protection	River Tummel, Scotland	Winterbottom & Gilvear (2000)	Aerial photographs and field data	1:24 000 to 1:5 000
Channel Modification	Artificial structures e.g. weirs Bank protection e.g. gabions, riprap	None known				
In-stream habitat	Flow type e.g. broken standing wave Biotope e.g. riffle, pool, run	Automated mapping of riffles, pools, eddy drop zones and runs.	Lamar River & Soda Butte Creek, USA	Marcus *et al.* (2003)	128-band hyperspectral imagery	1m pixels
Cross sections	Water depth Channel slope Bed morphology	Comparison of water depth estimates using a linear transform (Lyzenga, 1981) method and a log-transformed band ratio method	Soda Butte Creek, USA	Legleiter *et al.* (2004)	128-band & 4-band imagery	0.75-1m pixels
Channel substrate	Dominant bed material Quantitative measures e.g. D_{50}	Feature based automated classification of grain size and water depth.	Sainte-Marguerite River, Canada	Carbonneau *et al.* (2004)	Colour aerial photography	3cm pixels

It is also vital to consider catchment-scale processes and how the wider landscape structure may influence the physical, chemical and biological properties of in-stream habitat. If riparian and in-stream habitat walkover surveys are considered prohibitively expensive at the catchment-scale, then visiting every location within the landscape to determine whether it is having an impact on the aquatic environment is virtually impossible. To this end, a progressive engagement between remotely sensed data and mathematical models is enabling science to make statements about which locations in the landscape are likely to be causing habitat degradation without the need to visit those locations (Lane *et al.*, 2006). This is the focus of Chapter Five. Debate over the influence that landscape structure and land management activities at the catchment-scale may have over aquatic ecology via influences over flow regime, sediment dynamics and water quality has grown in recent years (Wang *et al.*, 2003). This interest has resulted in the inclusion of catchment land cover, determined through remote sensing, as a variable in several studies relating habitat to fish populations (e.g. Stauffer *et al.*, 2000; Wang *et al.*, 2003) and also within fisheries models used for stock size prediction (e.g. the Habscore model, Milner *et al.*, 1993). However, such studies and models typically only incorporate the proportion of land under particular cover types in the upslope contributing area and take no account of location, or of the hydrological processes such as hydrological connectivity operating within the catchment. Indeed the spatial distribution of hydrological processes such as infiltration, soil saturation, flow pathway and ultimately hydrological connectivity may be just as important, if not more important than the spatial distribution of land cover type in determining sensitivity to land management (Burt, 2001) and which parcels of land pose the greatest threat to in-stream habitat. For example, research has shown that not all locations within the landscape contribute equally to in-stream water quality degradation, even if they are under the same land use (e,g. Heathwaite *et al.*, 2000). Instead it is theorised that it is land use coupled with the ability for pollutants to be transported to the channel network that is crucial in determining water quality. The aim here, therefore, is to identify and to validate an approach which adopts explicit treatment of landscape location and the process of hydrological connectivity within its framework. Direct links with ecology will be made to investigate whether, in this case salmonid populations, are structured by catchment hydrological processes, firstly by hydrological connectivity alone and secondly by hydrological connectivity weighted by land cover.

To consider the case of catchment land management, nutrient export, water quality and ecology as one particular example, there are two major model types (empirical and physically based models) that have been applied within the scientific literature. Table 1.6 presents a classification of models found under these two types as described by Lane *et al.* (2006). Empirical models are generally the simplest form of model. Based primarily on the analysis of observations, they seek to characterise response from these data using statistical techniques (Merritt *et al.*, 2003). Their computational and data requirements are typically low making them advantageous at the catchment-scale. For example, Burt (2001) notes that export co-efficient models in particular have been used to model long-term nitrate loss from catchments with much success. However, such models are often criticised for ignoring the heterogeneity of catchment characteristics and the inherent non-linearities found within the catchment system. Further, and crucial to the aims of this research, they tend to be spatially aggregated or 'lumped' and therefore restricted to the extent by which they can represent the spatially distributed effects of location and hydrological connectivity (Burt, 2001; Lane *et al.*, 2006)

Table 1.6: *A simple classification of water quality models. Based on Lane et al. (2006)*

Broad-scale model type	**Specific modelling approach classified by Lane *et al.* (2006)**	**Description**	**Examples**
Empirical	Data inference	Infer diffuse pollution sources from detailed analysis of water quality data.	Nutrient budgeting (e.g. Cooper *et al.*, 2002)
Empirical	Transfer function modelling	Predict nutrient export on the basis of simple transfer functions driven by known nutrient inputs from fertiliser and manure applications and uptake under certain land cover types	Export co-efficient models (e.g. Johnes, 1996; Worrall and Burt, 1999)
Physically based	Land unit modelling	Model the physical, chemical and biological processes of nutrient cycling within individual land units to determine export.	(e.g. Priess *et al.*, 2001; Binder *et al.*, 2003)
Physically based	Land transfer modelling	Combine land unit modelling with a physically based sometimes dynamic treatment of how material is transferred across the landscape.	(e.g. Adams *et al.*, 1995; De Roo and Jetten, 1999)

Alternatively, physically-based models are based upon the solution of differential equations such as conservation of mass and momentum for describing stream flow and nutrient cycling within the catchment (Merritt *et al.*, 2003). Typically distributed in nature, they enable the effects of location and hydrological connectivity to be incorporated within their framework. However, as Lane *et al.*, (2006) note, this is only the case for land transfer models and even then they typically treat hydrological connectivity in a simplified way. With intensive computational and data requirements there is also typically insufficient capability to run models at the catchment-scale or to collect the field data required for parameterisation. This leads to parameters being determined through calibration against observed water quality and hydrometric data (Merritt *et al.*, 2003), which typically focuses on mean tendencies collected at coarse timescales. This tendency to use averaged data, attributing variability to "noise" caused by sampling and analytical errors, may lead to modellers and managers missing evidence of small scale information (Harris and Heathwaite, 2005). This results in the issue of equifinality, in that many combinations of parameters may result in an apparently successful calibration of the model, i.e. obtaining the correct results for the wrong reasons (Beven, 1989; Beven and Freer, 2001; Wheater, 2002, Heathwaite, 2003). Additionally, concerns have been raised over the uncertainty associated with model structures, which may over-simplify the complex heterogeneities of ecosystems due to partial evidence and incomplete understanding (Harris and Heathwaite, 2005). These limitations result in model output that is usually indicative rather than absolute (Sear *et al.*, 2003). As such, complex physically-based models may have no greater predictive power than simple empirical models. For example, Perrin *et al.*, (2001) rigorously compared model performance for 19 different model structures, finding that simple models frequently achieved similar levels of performance to more complex ones. However, they did also comment that there is a limit to model simplicity that is reached when the model fails to adequately account for observations. Simple models may also require greater calibration, as although they have fewer parameters, they can often have greater parameter sensitivity.

For these reasons, as has long been recognised in export coefficient modelling approaches, a more parsimonious approach to water quality modelling may be more cost effective whilst just as successful. However, there is still the need to incorporate explicit and spatially distributed treatment of hydrological connectivity at the catchment-scale within models. Lane *et al.*, (2006) propose that this problem can be solved through the use of risk-based prioritisation of critical source areas (CSAs) (Heathwaite *et al.*, 2000) within the landscape. This approach is similar to the transfer function approach. However, as Lane *et al.*, (2006, p243) state "rather than casting the export as a volume of material produced it is specified as a risk of material being produced, in relative terms compared with other units within the landscape". This risk can then be combined with explicit treatment of the risk of hydrological connectivity based on catchment topography and the location of land units, which will influence the dominant hydrological pathway, and the risk of coupling along flow lines which will determine the risk of delivery (Burt, 2001). Indeed, managers may not actually need absolute details; rather it is the spatial distribution of risk, with one location compared relatively to another that is necessary for targeting resources. A new environmental risk model, the *SCIMAP* model (Lane *et al.*, 2006) aimed at delivering this approach is currently under development.

Hence, Chapter Five evaluates the potential of the *SCIMAP* model for estimating the sensitivity of ecology, in this case salmonid populations, to the risk of connectivity present within the landscape. Model capability will be assessed by making direct links between salmonid populations and (1) hydrological connectivity alone and (2) hydrological connectivity weighted by land cover.

1.4.3 ***Objective (3):*** To use the data acquired under (2), to investigate hypotheses formulated through (1) regarding habitat controls and salmonid populations, and to discuss the results in the context of effective approaches to habitat restoration.

Objective (1) will identify a number of hypotheses regarding the relationships between salmonid populations and habitat controls at the in-stream, riparian and catchment-scale. Objective (2) will produce the data on habitat controls required to explore these hypotheses. The aim of Objective (3) is to develop an approach capable of integrating and analysing these data together with data on salmonid populations to test the hypotheses identified. The first stage to achieving this is the requirement to produce a spatially-structured hierarchical database integrating the habitat data acquired under (2) with data on salmonid populations. As discussed, habitat controls operating at any of the three scales (in-stream, riparian and catchment) have the potential to impact any point or reach within the channel network. Therefore, only those sites containing data at all three scales should be selected for analysis. Additionally, to evaluate the importance of investigation scale and location, sites must be also be associated with the various hierarchical scales of structure within the catchment (e.g. catchment, area and tributary). With powerful GIS systems specifically designed to manipulate, store and analyse large quantities of spatial data now widely available and accessible to practitioners, this type of database creation should be readily feasible. As the habitat data have been produced using different tools data will need to be spatially co-registered to facilitate extraction to a single database. The database should be capable of being interrogated both within the GIS itself and exported to statistical packages such as Microsoft Excel and SPSS for further analysis.

The second stage requires an approach capable of interrogating large, multivariate datasets. Researchers have previously used statistical methods such as linear regression (e.g. Bradford and Irvine, 2000), multiple regression analysis (Pess *et al.*, 2002; Coley, 2003), canonical correspondence analysis (CCA) (Wang *et al.*, 2003), and analysis of variance (ANOVA) (Stauffer *et al.*, 2000) to undertake similar studies. Multiple regression analysis has been selected here as it can establish whether a set of independent variables explains a proportion of the variance in a dependent variable at a significant level and, more importantly, can establish the relative predictive importance of the independent variables (Garson, 2006). This is important as it enables managers to identify which factors are mostly likely to be limiting salmonids and which restoration strategies may result in the greatest benefits to salmonid stocks.

Significant issues to consider with this type of analysis (Armstrong *et al.*, 2003) are the problems of spatial autocorrelation and collinearity that are likely to occur between habitat features. For example, in-stream variables such as gradient, substrate size, width, depth and velocity are all interrelated. Similarly, the location of various habitat pressures may also exhibit spatial autocorrelation. For example, livestock access is likely to go hand in hand with bank erosion due to stock poaching. If these variables are independently included within the statistical analysis, their effects may be double-counted resulting in selection of an end-model parameter suite that contains redundant parameters whilst not necessarily including those variables that exert most influence over population dynamics. Other researchers have also commented on this problem and have adopted procedures such as Principal Components Analysis (PCA) including transformed feature variables to alleviate the problem to some degree (Walters *et al.*, 2003). Analysis of spatial autocorrelation within the dataset should initially be undertaken at the catchment-scale. However, in order to explore the importance of scale in such studies, subsequent analysis should be undertaken where possible at an area and tributary level, to examine whether habitat controls relate to each other differently at different scales and in different locations.

The aim of this research was to develop an approach that would enable fisheries managers to prioritise restoration more effectively. Achieving this requires the results of the above analysis to be discussed in the context of developing strategies for restoration using the Eden catchment as a case study. As Lane *et al.* (2006) state "To ensure the most efficient deployment of mitigation effort it is important to focus upon those parts of a catchment where restoration is likely to give the greatest added value."

1.5 Summary of thesis structure

The structure of this thesis closely follows the order of the three objectives outlined in detail above.

Chapter One: This has presented an overview of the research, setting the context, discussing the specific case study approach and presenting detailed objectives.

Chapter Two: This will concentrate on Objective (1) and aims to provide a synthesis of current ecological, geomorphological and hydrological understanding of the in-stream, riparian and catchment-scale controls on salmonid habitat in order to identify gaps in current knowledge and identify hypotheses for investigation. Whilst this research is specifically concerned with the impact of in-stream, riparian and catchment-scale controls upon the freshwater habitat of salmonids, to set it within the wider context of salmonid research Chapter Two will first provide an overview of the alternative theories for decline and consider their relevance to the Eden's salmonid populations (Section 2.3). Discussion will then consider:

(1) Ecological theory of salmonid population regulation including stock recruitment, carrying capacity, density-dependent and density-independent mortality (Section 2.4);

(2) Ecological understanding of habitat (both physical and water quality) requirements and impacts of habitat degradation throughout the salmonid life-cycle, including hypotheses regarding habitat controls and reviews of research relating salmonid populations to these controls (Section 2.5); and

(3) Hydrological and geomorphological understanding of the processes which control in-stream conditions, including hydrological connectivity and the impact of land management upon in-stream morphology and catchment hydrological processes (Section 2.6).

The information provided within the above discussion will then be synthesised by developing a diagrammatic, conceptual model summarising current understanding and expert opinion regarding the in-stream, riparian and catchment-scale controls on salmonid habitat. Finally, the synthesis will be examined to identify a set of hypotheses regarding relationships between salmonid populations and habitat controls for investigation within Chapter Six.

Chapter Three: This thesis is heavily reliant on a large amount of spatial data, much of which has to be sourced from a variety of third parties. To this end, Chapter Three aims to identify the broad-scale data needs of this research, and to review the catchment-wide data sources that are available to fulfil these needs for the Eden catchment. This will include a review of fisheries data, digital topographic data, satellite imagery and aerial photography. The methods by which these data have been collected will be discussed together with an evaluation of their accuracy and the uncertainty involved in their use.

Chapter Four: The aims of Chapter Four are to identify and to develop new tools for the derivation of in-stream and riparian habitat data based on the data sources presented in Chapter Three, specifically 20cm digital aerial photography and the NEXTMap Britain 5m digital terrain model (DTM). Methodologies for the tools developed will be presented including the use of aerial photography to carry out a virtual walkover survey within GIS (Section 4.2); DTM processing to estimate channel slope and in-stream flow type (Section 4.4); and the automated classification of water depth and in-stream habitat from aerial photography (Section 4.3). The capability of these tools will then be evaluated: (1) in terms of their accuracy through the use of ground validation and established accuracy assessment procedures based on the error matrix (Congalton and Green, 1999); and (2), equally importantly, in terms of their practicality and cost compared with traditional walkover survey techniques.

Chapter Five: The aims of Chapter Five are to identify and to validate an approach which adopts explicit treatment of landscape location and the process of hydrological connectivity within its framework, making direct links with ecology to investigate whether, in this case salmonid populations, are structured by catchment hydrological processes: (1) by hydrological connectivity alone and; (2) by hydrological connectivity weighted by land cover. The chapter will begin with a general overview of hydrological modelling approaches, focusing in detail on the selected approach, the *SCIMAP* framework. The methodology of producing catchment risk maps of (1) hydrological connectivity and (2) hydrological connectivity weighted by land use will be presented together with the risk maps produced for the Eden catchment. Direct links through statistical analysis will then be made between salmonid populations and hydrological connectivity risk, in order to validate the technique and to assess whether salmonids are structured at the catchment-scale by catchment hydrological processes. The technique will also be evaluated in the context of an intensive spatial sample of water quality, and evidence from gravel siltation mapping.

Chapter Six: This chapter will concentrate on Objective (3), its aim being to use the data acquired in Chapters Three, Four and Five to investigate hypotheses identified in Chapter Two regarding habitat controls on salmonid populations. The chapter will first present the methodology of developing a spatially-structured hierarchical database of catchment-wide habitat data including GIS processing, co-registration of the data sources and integration with salmonid population data. This will be followed by statistical analysis. Analysis will be undertaken in three stages: (1) habitat controls will be related to salmonid fry data at the catchment-scale; (2) habitat controls will be related to salmonid fry data at an area and, where possible, tributary-scale; and (3) habitat controls will be related to salmonid parr data for an area of the Upper Eden catchment.

Chapter Seven: The results of the analysis undertaken in Chapter Six will be discussed in terms of the three hypotheses identified in Chapter Two and in the context of effective approaches to prioritising habitat restoration, using the Eden catchment as a case study example. This will include discussion of specific restorations strategies for the Eden catchment and the wider implications for fisheries managers undertaking catchment restoration for salmonids and fisheries in general.

Chapter Eight: This chapter will conclude the thesis by revisiting the aims and objectives of the thesis, summarising results in terms of their ability to fulfil the aims, discussing their implications for fisheries management, and highlighting areas for future research.

Chapter Two - Current understanding of salmonid habitat controls

2.1 Introduction

It is now widely acknowledged that assessing freshwater ecological integrity and developing sustainable and effective management policies requires a catchment or ecosystem approach (e.g. Petts, 2000). This should consider how policies and restoration strategies at one level within the ecosystem may impact upon functions at other levels in the system (Elliott, 2002; Karageorgis *et al.*, 2006) and, in so doing, be holistic. Such an analysis requires a mechanism for synthesising and integrating a large volume of data from across disciplines, within a common conceptual framework. A number of approaches and frameworks have been applied within environmental management for synthesising knowledge of environmental systems in order to inform decision making. These include environmental accounting which combines economic and environmental information to assess the contribution of the environment to the economy and the impact of the economy on the environment (e.g. Lange *et al.*, 2003); adaptive management approaches that accept uncertainty and experimentally test a range of alternative management approaches, refining them over time based on comparison of results (e.g. Gregory *et al.*, 2006); ecological risk analysis based on problem identification, analysis, risk characterisation and risk management (e.g. Stohlgren and Schnase, 2006); the 'Outcomes' approach (Olsen, 2003) in which policy implementation is linked to the behavioural and societal changes required, to enhance the environment; and SWOT (Strengths, Weaknesses, Opportunities, and Trends) analysis which evaluates different management strategies against each other (Pesonen *et al.*, 2001). Whilst all these techniques have their individual merits, one particular approach, the DPSIR (Drivers – Pressures – State – Impact – Response) approach has received considerable attention in recent years, especially from within the European Union (EU). This approach is now recognised as a powerful scoping framework for visualising complex environmental issues with regards to sustainable development and for linking science to the causes of environmental change (Karageorgis *et al.*, 2006; Borja *et al.*, 2006), or in this case linking the hydrological and geomorphological drivers and pressures at the riparian and catchment-scale to ecological impacts upon salmonid habitat and hence populations at the in-stream scale. It is therefore the approach that has been selected for use within this research.

Thus, the aim of this chapter is to use the DPSIR framework to help achieve Objective (1) of the thesis: *to review and to synthesise current understanding of in-stream, riparian and catchment-scale controls on freshwater salmonid habitat throughout the salmon and trout life-cycle.* This synthesis will then be used to select a suite of habitat controls from across all three scales for further investigation within this research together with hypotheses about how those controls may impact upon populations.

2.2 The DPSIR conceptual framework

The DPSIR approach originated within the OECD (Organisation for Economic Co-operation and Development) as the PSR (Pressure – State – Response) approach in the early 1990s (OECD, 1993) but has since been developed and extended to the DPSIR approach by the European Environmental Agency (EEA) with particular reference to integrating social and economic policy development with their environmental impacts. The approach has now been widely applied to a number of environmental management issues including, implementation of the Water Framework Directive (WFD) (e.g. Borja *et al.*, 2006), soil erosion risk assessment (e.g. Gobin *et al.*, 2004), coastal and marine management (e.g. Elliot, 2002; Karageorgis *et al.*, 2006), selection of climate change indicators (e.g. Donnelly *et al.*, 2004), and assessment of agricultural pressures upon water quality (e.g. Giupponi and Vladimirova, 2006). The main advantages of the DPSIR approach are that it is simple and generic. As seen from its varied applications, it can readily be adapted to varying and often complex issues including in this case to the identification of controls upon salmonid habitat. Figure 2.1 illustrates the DPSIR approach. Each of the terms in Figure 2.1 can be defined with respect to salmonids as follows:

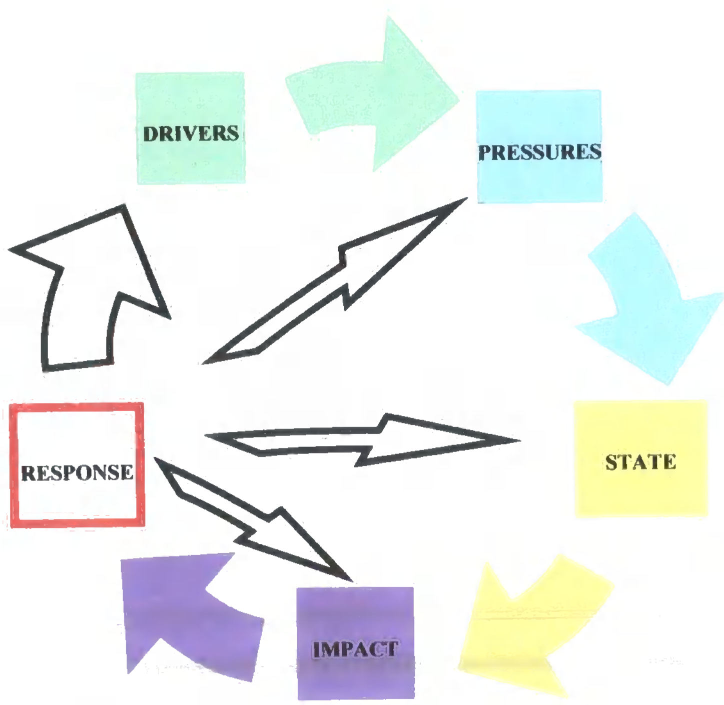


Figure 2.1 *The DPSIR framework*

Drivers typically relate to the social and economic policies and goals of governments, businesses and individuals that result in environmental changes, without regard to their specific impacts (Borja *et al.*, 2006). This includes population expansion, urban and industrial development, agricultural intensification, and exploitation of natural resources (e.g. fishing). Natural environmental conditions, such as topography, geology, soil type and climate can also be classed as 'Drivers' should they control the environment's sensitivity to change or in the case of this research control the environment's suitability for salmonids. Climate change is a unique 'Driver' as it is a natural phenomenon but one which is being influenced by anthropogenic greenhouse gas emissions.

Pressures represent the specific ways in which driving forces are expressed and by which ecosystems and their components are perturbed (Borja *et al.*, 2006), resulting in a change in the 'State' of the environment. This thesis is particularly focused on identifying those 'Pressures' which control the physical, chemical or biological state of the freshwater environment. This includes anthropogenic pressures such as catchment land use, river canalisation, sewage treatment and industrial discharges, road runoff, dam construction, land drainage, gravel extraction, and agricultural stock access to channel banks, together with natural pressures such as hydrological connectivity and rainfall intensity.

State refers to those components of the environment that are affected by 'Pressures'. In relation to the freshwater environment the 'State' is usually considered to comprise of hydraulic components such as flow regime, frequency and duration of inundation, depth and velocity; and abiotic components such as channel dynamics, bedform, channel slope, substrate composition, water temperature and water chemistry (Petts, 2000). These components then 'Impact' the biotic state of the environment, for example, the distribution and composition of macroinvertebrates, macrophytes, biological oxygen demand and dissolved oxygen concentrations. As predators, the distribution and abundance of salmonids not only depends directly upon the hydraulic and abiotic components of the ecosystem, but also on the biotic components. As such, from the perspective of salmonids, biotic factors are also part of the 'State' of the ecosystem, and through these salmonids may be indirectly impacted by hydraulic and abiotic components. This research therefore interprets 'State' as Atlantic salmon and brown trout requirements (hydraulic, abiotic and biotic) with the aim of specifically identifying those 'Pressures' and 'Drivers' that control salmonid habitat and hence salmonid populations.

Impact: relates to understanding the effects of altering the 'State' of the environment on a particular organism or ecosystem function of interest. Within general applications of the DPSIR framework, this has typically focused on impacts to human health, or ecosystem functions of social and economic importance such as flood risk. However, in terms of this research, 'Impact' will relate to understanding the mechanisms by which salmonid habitat and changes to that habitat may influence salmonid abundance and distribution.

Response: relates to the identification of strategies and policies such as, regulations and taxes which can remediate detrimental impacts on the 'State' or in this case to environmental strategies that can restore, rehabilitate and enhance salmonid habitat and hence salmon and trout populations. With regard to the various strategies employed there has been considerable debate regarding the precise definition of the terminology used. In the strictest sense, the term restoration is generally acknowledged to mean a radical attempt to recreate the structure and function of a system prior to disturbance or pressure (Cairns, 1991, cited in Wheaton, 2007; McDonald et al., 2004). Rehabilitation typically refers to the partial return to a pre-disturbance state through the recreation of certain, but not all, ecosystem functions, as a result of natural, social and economic restrictions upon a pure restoration process (Cairns, 1991, cited in Wheaton, 2007; McDonald et al., 2004). For example, irreversible changes to abiotic and biotic factors may have occurred preventing a return to former conditions even when pressures are removed, or features of cultural or historic importance (e.g. national monuments, infrastructure, high value agricultural land) may need to be preserved (McDonald et al., 2004). Enhancement refers to the alteration of a site to produce conditions that did not previously exist in order to accentuate one or more values of a site (Lewis, 1989 cited in US EPA, 2007). For example, flow deflectors, weirs and rubble mates may be used to promote habitat diversity within heavily engineered channels (O'Grady et al., (2002). In this case, the river is being modified to provide a function that is economically or socially desired without restoring true ecosystem function. Despite the precise definition of restoration it is often loosely applied to mean any variety of river management activities including rehabilitation, enhancement and creation (Brookes, 1996, cited in Wheaton, 2007). Throughout the thesis the term restoration is used in this broad sense unless specific reference is made to rehabilitation or enhancement.

Responses can be directed at any other part of the system (Borja *et al.*, 2006). In general, 'Responses' targeted at 'Drivers' should result in the most sustainable restoration or rehabilitation as they address the drivers of ecosystem function, but they are often the most difficult to implement involving high economic costs and widespread behavioural changes. In comparison those targeted at the 'State', typically enhancement schemes may be less sustainable and require on-going maintenance as 'Pressures' still remain. However, they are often cheaper in the short term and achieve more immediate benefits. As a result many traditional restoration 'Responses' are targeted at the 'State'.

In terms of the DPSIR framework the aim of this chapter is to review and synthesise current understanding and expert opinion regarding the in-stream, riparian and catchment-scale controls ('Drivers' and 'Pressures') of salmonid habitat ('State') and their subsequent 'Impact' on salmonid populations. Scientific understanding of the controls upon salmonid habitat falls primarily within three areas which can be incorporated within the DPSIR framework as follows:

1. Ecological research has focused upon the mechanisms by which natural populations are regulated including the impact of habitat availability on population dynamics. This equates to the 'Impact' section of the DSPIR framework.

2. Fisheries science has studied the habitat requirements of salmonids or in terms of DPSIR the optimum 'State' required to maximise their performance.

3. Hydrological and geomorphological research on the other hand has focused on the actions, variables and processes ('Drivers' and 'Pressures') which control in-stream hydraulic and abiotic 'State' conditions.

Collaboration between ecologists, fisheries scientists, hydrologists and geomorphologists has been slow to evolve. The benefits of collaboration are emerging, but are yet to be fully realised (Petts, 2000). The DPSIR framework focuses and encourages their integration, with the common link between ecological and hydrological/geomorphological research being the understanding of the 'State'. Biologists consider the state in terms of biotic habitat requirements or habitat niches whilst hydrologists and geomorphologists consider the state in terms of hydraulic and abiotic physical and chemical conditions. Synthesising a review of current understanding from across disciplines within the DPSIR framework should enable habitat controls ('Pressures') to be identified for further investigation that control abiotic factors which are ecologically relevant to Atlantic salmon and brown trout. It is also important to recognise that due to changing habitat requirements and mobility throughout the salmonid life-cycle (Armstrong *et al.*, 2003) different controls at different scales may be relevant at different stages of the life-cycle. In this respect the

'State' of the ecosystem may limit specific life-stages more than others. This has resulted in the widely adopted concept of "bottlenecks" within fisheries science (e.g. Summers *et al.*, 1996; Hendry, *et al.*, 2003). For example, if fry numbers are constrained by habitat then parr populations may be low regardless of habitat availability at the parr life-stage. It is important to accommodate this concept within the DPSIR framework as identifying population/habitat bottlenecks enables restoration strategies to be targeted towards particular life-stages where they may achieve maximum benefits. As such, habitat requirements ('State') will be discussed in terms of specific life-stages, principally, spawning, fry and parr.

The DPSIR framework is not a model. Whilst it may identify potential hypotheses regarding the causes and effects of habitat degradation, it will not model the dynamics and processes that link components (Karageorgis *et al.*, 2006). This must be done either by employing specific scientific models such as hydrological runoff models to represent processes or by identifying appropriate indicators or measures of drivers, pressures, state and impacts that can be the subject of statistical investigation. Identification of such models and indicators will be expanded upon in the proceeding chapters of this thesis. Additionally not all 'Drivers' and 'Pressures' may be applicable to the case study of the Eden catchment. Consequently, greater attention will be directed towards identifying and discussing those which are potentially applicable based on their likelihood of occurrence within the Eden catchment's landscape.

2.3 Hypotheses for salmonid decline: the wider context

This research is specifically concerned with the impact of in-stream, riparian and catchment-scale controls upon the freshwater habitat of salmonids and how this relates to salmonid abundance and distribution. However, habitat degradation is not the only hypothesis or potential driver for declining salmonid populations. In reality, it is most likely that the observed decline is due to the complex interaction of many factors. It is therefore important to recognise these alternative hypotheses and discuss their relevance in terms of the Eden catchment prior to development of the DPSIR model, in order to fully appreciate the role of freshwater habitat and pressures upon that habitat within the entire ecosystem function.

2.3.1 Exploitation

Of the many factors implicated in the decline of salmonid stocks, exploitation is the only activity that involves deliberately killing fish. It is therefore unsurprising that management of fishing levels receives considerable attention, as it is presumed that reductions in exploitation pressure will achieve the most direct and immediate improvement in stocks (Potter *et al.*, 2003). Exploitation can be both commercial, involving fisheries on the high seas, coastal waters and estuaries, and recreational within the freshwater environment by rod and line angling. Public awareness regarding the issue of exploitation was particularly raised following the dramatic collapse of another species, the Northern cod (*Gadus morhua*) fishery off Newfoundland and Labrador (Myers *et al.*, 1997). This led to concerns being raised about other major fisheries including Atlantic salmon.

Studies of returning Atlantic salmon have suggested that up to 70% of returning fish may be taken by commercial marine fisheries (Dempson *et al.*, 2004). Such figures have led to the closure or downscaling of several fisheries. However, studies investigating the impact of this have found conflicting results. For example, following the regulation of drift netting in the River Usk estuary, Wales in 1992, there was a dramatic rise in the rod catches of Atlantic salmon on the River Usk (Bowker *et al.*, 1998). However, ten years after the closure of the Newfoundland commercial Atlantic salmon fishery in 1992, which took an average of 905 t yr^{-1}, little difference in smolt abundance and even a decline in marine survival was observed (Dempson *et al.*, 2004). They attributed this to climatic fluctuations in the North Atlantic or an increase in predators throughout the period of study, and concluded that without the closure of the fishery the resource would now be in a particularly depressed state. Alternatively, Potter *et al.* (2003) comment that, whilst exploitation reduces the number of available spawners and therefore eggs, there should be less competition between juveniles and so a greater proportion should survive. They suggest that fisheries can sustain relatively high exploitation rates and give the example of the productive Northern Esk, Scotland where over 50% of the returning fish are exploited. However, they do add that there will be a critical exploitation rate above which the population's productive capacity will be impaired potentially leading to collapse. They discuss the use of conservation limits and management targets to set threshold spawning requirements for each stock which can be monitored enabling decisions regarding harvesting levels to be managed. This is no easy task since many stocks are exploited in common fisheries such as West Greenland and Faeroes.

In addition to impacts on salmonid abundance, exploitation has also been linked to changes in population structure (Almodova and Nicola, 2004) phenotypic behaviour and genetics (Consuegra *et al.*, 2005). Evidence suggests that exploitation is often selective, leading to increased pressure and higher mortality for particular components of the population. Adult Atlantic salmon that enter the river in spring are often dominated by large multi-sea-winter females. Fish that enter later in the year are dominated by smaller male grilse. It has been noted (Consuega *et al.*, 2005), that the larger spring run fish, prized by anglers, have been particularly exploited in many areas, leading to detrimental genetic and phenotypic responses such as reductions in size, sea-age and longevity of fish and delayed entry into the river that may reduce the adaptive ability of these populations to cope with stress. Similar observations have been noted for brown trout (Almodova and Nicola, 2004), where selective exploitation of large fish has led to decreases in the mean age, and age diversity of populations, resulting in reductions in the available breeding stock and population fecundity. Such findings have led to the introduction of legislation aimed at protecting vulnerable components of populations. For example, in April 1999 the Environment Agency in England and Wales introduced a national bye-law aimed at protecting the multi-sea-winter spring running salmon. It states that

"Any person who removes any live or dead salmon taken by rod and line from any waters or banks without the previous written authority of the Agency before the 16th day of June in any calendar year shall be guilty of an offence. This Byelaw shall not apply to any person who lawfully takes a salmon and returns it immediately to the water with the least possible injury".

The use of slot limits that determine the size of fish that may be taken by anglers has also been advocated as a method of protecting vulnerable and essential components of the population (Nordwall *et al.*, 2000; Almodova and Nicola, 2004). However, the use of such management techniques must be population-specific as different populations have different genetic characteristics exhibiting different growth rates and different maturation ages. Legislation that only protects one population may be detrimental to another.

In terms of the Eden catchment, Table 2.1 presents a summary of the national and regional bye-laws governing rod and line angling. These are set and regulated by the Environment Agency which is responsible for managing the freshwater fishery.

Table 2.1: *National and regional bye-laws governing rod and line angling in the River Eden catchment, Cumbria (Environment Agency, 2005.)*

Species	Closed season	Catch and release	Size limits
Atlantic salmon	15th October – 14th January	15th January – 16th June	Only adult fish may be taken
Non-migratory brown trout	1st October – 14th March	At the angler's or fishery owner's discretion	No fish under 200mm in length may be taken
Migratory trout	1st October – 31st March	At the angler's or fishery owner's discretion	No fish under 300mm in length may be taken

The issue of exploitation is complex involving both the conservation of species and allocation of surplus resources to achieve the greatest socio-economic benefits. To ensure continued and sustainable utilisation of the resource, fisheries must be managed in such a way as to protect the productive capacity of stocks and maintain biological diversity (Potter *et al.*, 2003); this is often associated with improving freshwater habitats to maximise production.

2.3.2 Aquaculture

At the same time as wild stocks of Atlantic salmon and brown trout decline and tighter regulations are imposed on exploitation, aquaculture (fish farming) is booming in order to meet the increasing demand for fish. Today, over 94% of all adult Atlantic salmon are in aquaculture (Gross, 1998), an industry that has seen major intensification over the last two decades (Read and Fernandes, 2003). However, whilst this may appear to reduce pressure on exploitation of wild stocks, it comes with its own issues that have been suggested as contributory causes to the current decline in wild stocks. Aquaculture has been linked to: increased instances of disease in wild stocks (e.g. McVicar, 1997; Bjorn *et al.*, 2001); degradation of surrounding water quality due to discharges of effluent containing waste feed, faeces, medications and pesticides (Read and Fernandes, 2003); and genetic weakening of wild stocks through interbreeding of escaped and stocked farmed fish (e.g. Gross, 1998).

In particular, the movement of live fish for farming or re-stocking has been associated with the transfer of pathogens from one region where fish may have natural resistance to another where the fish have no natural resistance. This can lead to devastating results should the exotic infected fish escape or be stocked in new waters. One of the most notable cases of disease transfer was of *Gyrodactylus salaris*, from Baltic salmon to Atlantic salmon, devastating wild stocks in Norway (McVicar, 1997). High instances of infection within farmed fish have also been blamed upon the high densities and environmental conditions under which fish are kept, and epidemics within fish farms have been linked to outbreaks of infection in nearby wild stocks. For example, sea trout feeding in the vicinity of a salmon farm in Northern Norway were found to have 100-200 salmon lice (*Lepeophtheirus salmonis*) per fish compared with only 10 lice per fish in an area remote from farming activity (Bjorn *et al.*, 2001). The development or evolutionary forces operating within the aquaculture environment are very different to those operating in the wild, and it has been suggested (Gross, 1998) that this is leading to the development of two very different biologies. Farmed fish have relatively restricted genetic origins compared to wild fish and are therefore less genetically diverse. This has led to concerns being raised about the impact of reared fish interbreeding with wild fish on the genetic health of wild stocks and their subsequent resilience to natural stresses and reproductive viability (Thompson *et al.*, 1995). However, the industry is heavily regulated in terms of fish movements, farming conditions, and water quality. Within Europe regulation comes under the remit of the Common Fisheries Policy (CFP), but there are many other national and international laws which also apply (see Read and Fernandes, 2003, for a review).

Aquaculture within the Eden catchment is limited with only one major fish farm that operates two sites. One is located at the foot of Haweswater reservoir (Figure 2.2.) and produces Atlantic salmon smolts. The other is a hatchery at Holmwrangle, Armathwaite. The company states that their sites are "totally isolated from wild stocks and that all water leaving the sites is cleaned to a high standard removing 95% of the solids down to 60 microns before passing through a final Bio pond prior to discharging" (www.lakelandsmolts.co.uk). A number of smaller fish farms including Sockbridge Mill trout farm at Eamont Bridge and the Environment Agency's hatchery at Warwick Bridge have all recently closed. However, stocking of the river, tarns and reservoirs with adult brown trout from outside the catchment by angling associations does occur.

Figure 2.2: *The Lakeland Group fish farm at Burnbanks, Haweswater. (Photo source: www.lakelandsmolt.co.uk)*

2.3.3 Disease

Infectious disease outbreaks (epizootics) are another factor which can result in a reduction in salmonid population size and which can occur dramatically over a relatively short time period. As discussed above, the introduction of exotic fish diseases through movements of live fish can result in severe consequences for populations. *Gyrodactylus salaris* is currently considered the greatest exotic fish disease threat to the UK with the movement of rainbow trout identified as the likeliest method of transmission (Peeler *et al.*, 2004). However, it is not just the introduction of exotic diseases which pose a threat. There are many other fish diseases that are naturally endemic within populations and the environments in which they survive, but which only result in outbreaks at specific times under specific conditions. In the UK one such disease is Ulcerative Dermal Necrosis (UDN), never reported in farmed fish. It is a condition that predominantly occurs in adult Atlantic salmon and sea-trout but which may also affect brown trout to some degree. The first recorded outbreak of UDN occurred in 1877, after which it largely disappeared until the 1960s when an outbreak originating in south west Ireland spread to most Irish and British rivers (Roberts, 1993), including a major outbreak in the River Eden in 1966, which particularly affected the multi-sea-winter, spring run salmon. A number of authors have noted that epizootics of endemic diseases coincide with very high wild stock numbers such as occurred in the UK in the 1960s, and suggest that this may be no coincidence, with high densities aiding the spread of disease (George, 1991; Roberts, 1993).

It has also been hypothesised that changes in environmental conditions (such as increased pollution) may lead to greater infection rates by altering the host's susceptibility to disease. For example, juvenile Pacific salmon (*Oncorhynchus spp.*) have been shown to bioaccumulate chlorinated and aromatic hydrocarbons that can result in immuno-suppression and increased disease susceptibility (Arkoosh *et al.*, 1998). This again raises the importance of protecting salmonid habitat, both physical and chemical, as maintaining a high quality environment should reduce stress on fish, improve population health and therefore ultimately reduce the risk of disease.

2.3.4 Predation

Within the commercial and recreational fisheries communities there is particular concern over the impact of predation upon Atlantic salmon and brown trout populations especially from pisciverous (fish-eating) birds including, in the UK, great cormorants (*Phalacrocorax carbo*) and goosanders (*Mergus merganser*). A rapid increase in the number of these birds roosting near and feeding within freshwater habitats has been observed since the 1980s especially during the winter months (Defra, 2004). Several studies have examined the stomach contents of birds (e.g. Stewart *et al.*, 2005) or tracked the fate of radio-tagged fish (e.g. Dieperink *et al.*, 2002) confirming that pisciverous birds can take considerable quantities of salmonids.

Fish have been observed to be particularly vulnerable to predation from birds under a number of circumstances. First, although normally territorial, during smoltification salmonids aggregate and migrate to sea in large numbers with particularly high densities occurring in areas where fish are funnelled through narrow channels, for example, fish passes around weirs and dams. Second, vulnerability may be high in open water bodies such as estuaries (Dieperink *et al.*, 2002), lakes and reservoirs (Jepsen *et al.*, 1998), where cover from predators in sparse. Associated with this, several researchers have hypothesised that the stocking of fish (especially adult fish) for angling purposes encourages predation due to increased fish densities, which may cancel out the effects of stocking on fishery performance (Stewart *et al.*, 2005). Similarly changes in land management such as agricultural intensification and urbanisation which reduce riparian vegetation and remove overhead cover may increase the fishing efficiency of predators resulting in a decline in stocks.

The question as to whether predation actually causes salmonid population decline remains unanswered and requires further studies involving accurate estimates of populations and predator-prey relationships (Carss and Marquiss, 1999). It is important to note that it is not only birds that prey upon salmonids. Researchers have also found that species such as pike (e.g. Jepsen *et al.*, 1998), otter (*Lutra lutra*) (Jacobsen, 2005) and seals take significant volumes of salmonids. Whilst all these species may prey on salmonids, predation is a natural component of ecosystem function. However, should some change such as, a reduction in the availability of cover or the migration of birds into a new geographical area as a result of climate change, disturb the natural predator-prey equilibrium, severe consequences for the prey species may be observed.

Within the Eden catchment pisciverous bird surveys were undertaken biannually by the River Eden District Fisheries Association (REDFA) between 2003 and 2005 (Table 2.2). They observed the greatest number of birds during March immediately prior to the Atlantic salmon smolt run when fish are most vulnerable, reporting higher numbers (5.71 times more goosanders and 1.41 times more cormorants) than the Breeding Birds of Cumbria tetrad survey 1997-2001 (Pers. Comm. J.Brown). However, the reason for an increase in goosanders and cormorants and their impact upon fish populations in the River Eden remains unknown.

Table 2.2: *Biannual pisciverous bird survey counts 2003-2005 (REDFA, 2006)*

	Oct-03	Mar-04	Oct-04	Mar-05	Dec-05
Cormorant	203	230	68	234	95
Goosander	302	349	150	357	193

2.3.5 Invasive species

The implications of introducing reared fish of the same species as a consequence of aquaculture or stocking have already been discussed, both in terms of disease transmission and genetic alterations to wild stocks through interbreeding. Introductions can also include non-indigenous species resulting in further pressures on native populations. This includes direct pressures such as competition for habitat and food, or predation of native species. There are also indirect pressures such as trophic cascades or even changes in ecosystem functioning, including nutrient cycling and energy transfer (Simon and Townsend, 2003). Exotic species are often highly successful in their new environments and cases of native species expatriation are common. Examples include the escape or release of American mink (*Mustela vison*) in Britain and their decimation of native water vole (*Arvicola terrestris)* populations; the impact of zebra mussels (*Dreissena polymorpha*) on freshwater pearl mussels (*Margaritifera margaritifera*) in North America (Ricciardi, 2003), and following their introduction through live baiting by pike anglers, predation of vendace (*Coregonus albula)* eggs by ruffe (*Gymnocephalus cernuus)* in Bassenswaite Lake, Cumbria, UK (Winfield and Durie, 2004). Reasons for the success of these invasive species may include: (1) that they are simply more effective competitors having evolved in a more competitive environment; and (2) that they are free from natural predators in their new environment (Allendorf and Lundquist, 2003). In terms of Atlantic salmon and brown trout, whilst a review of the scientific literature identified a number of studies investigating their competition with other salmonid species such as rainbow trout (*Oncorhynchus mykiss*) and grayling (*Thymallus thymallus*) (e.g. Degerman *et al.*, 2000; Scott and Irvine, 2000) few examples of severe negative impacts due to invasive non-salmonid species could be found. Interestingly, one non-fish species, the signal crayfish (*Pacifastacus leniusculus*), introduced to UK rivers from North America, has been shown to out-compete juvenile Atlantic salmon for over-wintering habitat, leaving them vulnerable to predation (Griffiths *et al.*, 2004). Conversely, there were more references to salmonids, brown trout in particular, acting as an invasive species, following its introduction as a sport fish into many countries including New Zealand and Australia (e.g. Simon and Townsend, 2003).

In terms of the Eden catchment, whilst several non-indigenous coarse fish species (e.g. pike, perch, chub and dace) and grayling have been introduced, many of these species have been present for hundreds of years becoming naturalised and developing their own particular niches, co-existing with Atlantic salmon and brown trout without negative effects. For example, the moats of Carlisle castle were stocked with pike as early as 1298, whilst grayling were introduced to the

Eden in 1883 (Pers. Comm. C.Bowman). However, in an attempt to help prevent further introductions that could lead to negative consequences, the Environment Agency introduced in 2002 a byelaw stating that the *"use of any dead or alive freshwater fish, salmonids or eels as bait is prohibited"* on or in Ullswater, Haweswater, Brotherswater and Red Tarn, all still waters within the Eden catchment (Environment Agency, 2005). In fact the invasive species causing the greatest threat to Atlantic salmon and brown trout in the Eden catchment may not be a fish species at all but a plant species, Himalayan balsam (*Impatiens glandulifera*) which has extensively colonised the lower catchment's riparian zone resulting in increased bank erosion and potentially increased siltation of spawning gravels.

2.3.6 Climatic variability and change

Climatic conditions are believed to exert a major influence over aquatic ecosystems in general and salmonid populations in particular due to impacts on both thermal and flow regimes. For example, Martin and Mitchell (1985) observed variations in the timing at which Atlantic salmon returned to spawn in the River Dee, Scotland during the period 1877-1972 associated with fluctuations in sea surface temperature (SST) in the sub-Arctic sea. Lower SSTs were correlated with reduced numbers of multi-sea-winter (MSW) fish and increased numbers of grilse (1-sea-winter) fish, and vice versa. George (1991) noted similar observations for other Scottish rivers, commenting that as a grilse-dominated period declines, the following MSW period lags behind, resulting in a period of low returning numbers from all age classes. However, as a MSW-dominated period declines, the rise of the grilse period is concurrent yielding large runs of both age classes for a number of years.

Cyclic variations in thermal and flow regimes such as sea surface temperature have since been associated with global climate variations and large scale fluctuations in atmospheric phenomenon such as the North Atlantic Oscillation Index (NAOI), and the El Niño Southern Oscillation (ENSO), (e.g. Trenberth and Hurrell, 1994; Hurrell and VanLoon, 1997; Rodwell *et al.*, 1999). These phenomena have also now been statistically linked to fluctuations in the dynamics of salmonid populations. For example, Boylan and Adams (2006) directly related cyclic variability in the abundance of returning Atlantic salmon to the River Foyle, Ireland to the NAOI in winter. Climatic influences are not restricted to the marine environment: inter-annual variations in the date of trout fry emergence for a small Lake District beck has also been related to cyclic fluctuations in the NAOI and stream temperature (Elliot *et al.*, 2000). Further, the effects of smaller scale, extreme weather patterns such as droughts and floods have also been shown to be important and influence salmonid population dynamics, behaviour and survival on an inter-annual basis, from estuarine conditions and upstream migration (Solomon and Sambrook, 2004) to redd stability and egg survival (Brown 2006a).

This sensitivity to climatic variations has lead to much debate and concern over the potential effects of anthropogenic induced climate change, particularly global warming due to increased greenhouse gas emissions, upon salmonid population dynamics in the future (e.g. Scott and Poynter, 1991; Swansberg *et al.*, 2002; Borgstrom and Museth, 2005; Boylan and Adams, 2006). Some researchers consider that future climate changes will and are already being detrimental to stocks. Juvenile Atlantic salmon have been observed to be getting smaller since 1971 in Eastern Canada, a trend that is inversely related to increasing spring air and water temperatures (Swansberg *et al.*, 2002). The authors suggest this is evidence of current climate change and that future predicted temperature rises may reduce growth further and ultimately diminish the overall productivity of Atlantic salmon in this region. Others are less certain about the future. For example, Borgstrom and Museth (2005) consider that juvenile brown trout recruitment in Norwegian mountain streams may be adversely affected by predicted increases in winter precipitation but that predicted increases in summer temperatures may increase recruitment.

In terms of the Eden catchment, North West England is predicted to experience increased seasonality, with warmer drier summers and wetter winters (Table 2.3). Short high intensity storms are likely to be more common especially during the summer and spells of extreme weather such as droughts and flooding are expected to occur more frequently.

Table 2.3: *Plausible changes in the climate of North West England by the 2050's as simulated by HADCM3*[3].

Climate Variable	Indicated change
Temperature	Annual temperature averaged across the UK may rise by 2-3.5°C depending on the scenario adopted. In the North West predicted summer temperature rises are between 0.8-2°C. The number of days below freezing may be reduced by up to 65%.
Winter precipitation	Winter precipitation is predicted to increase by as much as 6-13%. Flooding may become more frequent.
Summer precipitation	Summers will become drier with a reduction of up to 10% in rainfall. However, summer storms are predicted to become more intense with an increase in convective rainfall.
Seasonality	The contrast between winter and summer climate will increase under all scenarios.
River Flows	Seasonally river flows in the North West may show an increase of 5-15% in winter with little difference expected in summer.

(Based on Shackley *et al.*, 1998).

[3] Percentage changes are relative to the mean of the standard 1961-1990 baseline period.

Not only are these changes likely to have direct effects on salmonids via changes in thermal and flow regimes but also indirect effects. For example, climate changes are predicted to result in new opportunities for agriculture such as extended stocking periods and higher productivity of grasslands (Shackley *et al.*, 1998). The expansion of crops such as maize and winter cereals is already being seen within the Eden catchment. Such changes will have implications for water quality and erosion both from banks and catchment sources, and ultimately for freshwater ecology. Climate change has also been linked to predicted increases in coarse sediment delivery rates with implications for in-stream conditions and the frequency of flooding (Lane *et al.*, 2007).

What is clear is that climate variations do impact upon salmonid populations. What is less clear is precisely how future climate may change and how these changes may affect salmonid populations. What appears likely is that the Eden catchment will experience increased seasonality with more extreme weather conditions occurring more frequently. This may put pressure upon current salmonid populations. If such changes do occur, the availability of refugia habitat structures will become even more important. Additionally, it is important to recognise the role that land and water resource management strategies (e.g. land drainage, abstraction, deforestation) play in determining the environment's sensitivity and resilience to climate change and therefore the magnitude of changes that salmonids will have to adapt to and cope with. This further emphasises the importance of habitat in managing salmonid restoration.

2.3.7 Interactions between driving forces

The above discussion has touched upon the diversity and complexity of hypotheses associated with the decline of salmonid populations outside the issue of habitat degradation. It is unlikely that any one of these hypotheses or that habitat degradation alone can explain salmonid population dynamics. Rather, there are many complex interactions between factors and indeed many remaining unknowns that will ultimately determine salmonid population distribution and abundance. For example, the issues of exploitation and aquaculture are closely intertwined, both driven by the growing consumer demand for fish. Climate change has been associated with an increase in the incidence of invasive species observations as species ranges expand or contract leading to knock-on effects in terms of predation.

What has become particularly apparent through the above review is the link between many of the hypotheses discussed and the importance of habitat. These interactions are important to appreciate if the impacts of habitat management are to be fully understood. For example, Potter *et al.*, (2003) commented that for sustainable exploitation of resources, fisheries must be managed in such a way as to protect the productive capacity of stocks and maintain biological diversity. If habitat restoration enhances the productive capacity of a stock, exploitation rates will be sustainable at a higher level. In contrast, the pressure of exploitation will be felt to the greatest extent in degraded habitat regions. Pollution has been linked to reduced immunity and increased instances of disease (Arkoosh *et al.*, 1998); a cleaner environment will not only support greater numbers of fish but produce healthier individuals more able to survive extreme climatic events and long migrations resulting in higher longevity. Predation is closely linked to the ease with which prey can be caught. Habitat alterations that reduce cover such as tree clearance or bank reinforcement may increase the impact of predation. Habitat has also been shown to influence sensitivity to climate change. Populations within pristine, heterogeneous habitats with abundant refugia will be more capable of withstanding increased instances of extreme events such as floods and droughts than populations in degraded habitats.

The remainder of this review focuses on current understanding of salmonid habitat and its controls at the in-stream, riparian and catchment-scale within a DPSIR framework. This begins with 'Impacts' and current ecological understanding of population dynamics.

2.4 Ecological understanding of salmonid population dynamics

Most salmonid populations are naturally highly variable responding to changes in their environment. Understanding the mechanisms by which populations interact with their natural environment and respond to changes is important to help understand how they may respond to management interventions that aim to manipulate environmental conditions (Milner *et al.*, 2003). In terms of the DPSIR framework this refers to 'Impacts'.

It is well established that a given environment can only accommodate a certain number of fish and this is termed the "carrying capacity" (Summers *et al.*, 1996). The carrying capacity of any environment varies depending upon the species and life-stage of interest, territory and food availability and the time of year. There are two major theories regarding the mechanisms by which carrying capacity influences population abundance: (1) population regulation; and (2) population limitation, both of which are important to consider if the impacts of habitat degradation and habitat restoration strategies aimed at increasing carrying capacity are to be fully appreciated.

2.4.1 Theories of population regulation and population limitation

Population regulation is by far the dominant theory within the ecological community based upon the premises that: (1) populations are persistent and relatively stable over time fluctuating around a mean abundance; and (2) that no population can increase without limit, because that would ultimately mean to use up all resources, starve and become extinct (Elliott, 1994; and Milner *et al.*, 2003). Instead, the theory argues that some mechanism must actively act to regulate populations keeping them below the maximum carrying capacity of the environment, thereby preserving resources and allowing populations to persist. These mechanisms are theorised to intensify as population density increases and relax as it falls (Berryman *et al.*, 2002). This is more commonly known as "density-dependent mortality" and is first attributed to Nicholson (1933). The regulating forces proposed by Nicholson consist primarily of intraspecific competition for resources such as food and territory but also predation and parasites (Elliott, 1994). As described by Milner *et al.* (2003) at low spawning densities, following emergence from the redd, competition for resources is limited between young fry and survival high. However, as spawning numbers increase, so does competition following emergence, resulting in lower fry survival rates (Figure 2.3). Section 2.3.4 has already discussed the potential relationship between predation and population density, with increased predation observed following the introduction of stocked fish and section 2.3.2 discussed the increased incidence of parasitic infection within the high density aquaculture environment. These are all examples of negative density-dependence, in that the probability of survival decreases with increasing density. However, processes may also act positively so that the probability of survival increases with density (Milner *et al.*, 2003). For example the shoaling behaviour of smolts may act to reduce the probability of mortality from predation.

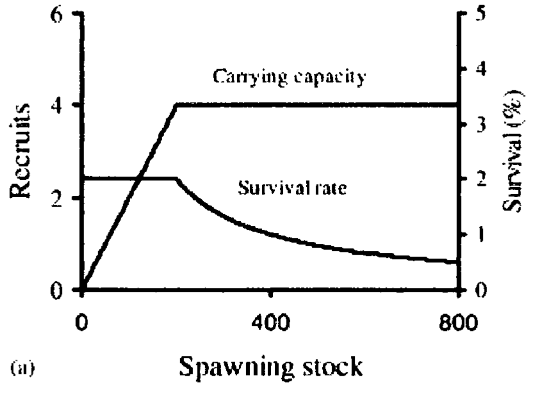


Figure 2.3: *Diagrammatic representation of recruitment constrained by carrying capacity, showing survival rate (% egg to recruit) changing with spawning stock (reproduced from Milner et al., 2003).*

Population regulation theory does not just include density-dependent factors, Nicholson (1933) also referred to density-independent factors which can cause mortality at any life-stage regardless of density. For example, pollution from fertilisers and pesticides may cause mortality through direct toxicity, siltation of redds through asphyxiation, or egg washout/desiccation at high/low flows through injury. These factors are thought to often mask the effects of density-dependent mortality due to their extensive impact which is often unpredictable (Milner *et al.*, 2003). Habitat availability and quality can influence the severity and probability of such events by determining an environment's resilience or sensitivity. For example, the presence of riparian buffer zones may increase resilience and reduce the impact of diffuse pollution. A resilient environment is likely to support more fish than a sensitive environment and therefore have a higher carrying capacity.

Whilst the theory of population regulation is well established, it is not without controversy. There are other researchers who are highly critical, particularly of the role of density-dependence, offering instead the theory of population limitation. Initiated by Andrewartha and Birch (1954), the theory of population limitation considers that abundance is passively limited by the inability of the environment to support all individuals (White, 2001), emphasising that population fluctuations are the result of fluctuations in environmental conditions and that there is no need to look any further for explanatory mechanisms (Elliott, 1994). Instead, they suggest it is more important to focus on identifying the limiting resource. Supporters of the limitation theory argue that the observation of population stability which drives population regulation is incorrect and that the reason populations remain below carrying capacity is not the result of competition or predation but because not all resources are easily accessible (White, 2001). For example, if prey became more accessible through the removal of cover, then the predators may increase until they destroy all the prey and ultimately starve. Population limitation theory does not deny that competition for resources occurs but proposes that it is a consequence not a cause, and that it is the environment that ultimately controls how many organisms survive, competition simply determines which ones. Nor does it reject predation, but instead argues that those fish caught represent the starving, sick, old or injured who would have died soon anyway (White, 2001). If environmental changes or events result in less food, cover, injury, sickness and more vulnerable fish, then increases in predation and competition are likely to follow, but in their absence abundance would still decline.

Others argue that there is really little difference between the two theories and that it is more a matter of perspective, with density-dependent mortality a mechanism that may or may not occur depending upon circumstances (Berryman *et al.*, 2002). Using the example of flooding in an environment with few refuges, Berryman *et al.* (2002) suggest supporters of regulation theory will observe density-dependent competition for a limited resource (refuges) with a density-independent factor (flooding) being the agent of mortality; whilst supporters of limitation theory will see passive limitation resulting from the inability of the environment (lack of refuges) to support populations in the event of a sudden environmental change (flooding). One focuses on the mechanism limiting abundance whilst the other focuses on the environmental condition setting the limit. Within the DPSIR framework this could be likened to the difference between 'State' and 'Impact'. It may also be unsurprising, as Elliott (1994) comments, that Nicholson worked in favourable controlled laboratory environments with stable populations close to their equilibrium, whilst Andrewatha and Birch worked in unfavourable environments where conditions and populations fluctuated widely.

2.4.2 Stock recruitment, self thinning and habitat bottlenecks

In addition to considering population controls at individual life-stages, it is also necessary to understand the connections between life-stages and principles of stock recruitment which determine how many fish at one life-stage will survive over time to develop into the next. Even in a system with abundant habitat, it is likely that the number of fish present will decline over time. It has been theorised that this is because as juvenile Atlantic salmon and brown trout grow the resources they require in terms of space and food increase. Therefore, if resources remain constant, competition will increase as fish grow. This process is known as self-thinning (Milner *et al.*, 2003). Elliott (1994), notes that this process is well documented for plants but that evidence for or against self-thinning in mobile species such as salmonids is limited. The fact that salmonids are mobile enables them to disperse and migrate in order to find new sources of food and larger territories as their requirements increase. Declining numbers may therefore represent emigration and dispersal not mortality. Further, a reduction in fish numbers over time may simply be the result of density-independent mortality events (Milner *et al.*, 2003). It is not just the amount of habitat (space and food) that changes throughout the salmonid life-cycle, different stages also have different types of requirement in terms of water depth, velocity, substrate and cover. Therefore the ability to progress to the next life-stage also depends on the ability to meet these changing requirements. If fish are unable to do this at any life-stage, numbers will either be regulated or limited resulting in a reduced number of fish at the next life-stage. The particular life-stage or habitat type at which this occurs is known as a 'habitat bottleneck'. An alternative view is that due to different habitat requirements at different life-stages a single environment will offer different potential for mortality to different life-stages (Summers *et al.*, 1996).

Habitat bottlenecks and high mortality are most commonly observed at the fry life-stage for a short period of time following emergence from the redd (Milner *et al.*, 2003; Einum and Nislow, 2005). This is intrinsically linked to their mobility and dispersal capability. As discussed in Chapter One (Section 1.2.2), fry are the least mobile life-stage, and have been observed to remain within hundreds of metres of their redd. Dispersal requires energy but at the same time it reduces feeding opportunities and leaves fish more vulnerable to predation (Einum and Nislow, 2005). Therefore, the further a fish is required to disperse, the less likely it is to survive. These effects are most acute at the fry life-stage, when an under-developed swimming capability increases energy expenditure and time exposed to predators per unit distance moved compared with older fish. Subsequently, fry are least able to disperse to avoid competition or find adequate habitat if that in their immediate vicinity is unsuitable or already occupied, instead dying in-situ. In this respect fry may be more sensitive to locally distributed habitat pressures and controls than older life-stages. Parr and adult fish are more mobile and are therefore more able to disperse to find suitable habitat and avoid competition. This has lead to the suggestion that parr and adult populations may be regulated or limited more by density-independent factors such as poor water quality than density-dependent ones (Milner *et al.*, 2003). However, even older life-stages have a limit to the range of their mobility and if suitable habitats are not available within that spatial range, populations may also be restricted by habitat bottlenecks at older life-stages, as observed by Elliot and Hurley, (1998).

Overall the more heterogeneous and pristine the environment, the higher the probability that fish will find suitable habitat within their spatial range. For example, Johnston *et al.* (2006) monitored the daily movements of salmon parr using PIT tags from July to November observing shorter daily movements in complex habitats. However, as spatial range varies with life-stage, it may be the case that fry are impacted more by pressures localised in extent, requiring a more heterogeneous and complex environment than parr that are limited by factors which are more widespread (endemic) throughout their environment as a whole. This has resulted in development of Hypothesis (1).

Hypothesis (1) Relationships between habitat and salmonid abundance/distribution are structured by life-stage according to the level of mobility and potential for dispersal at each life-stage.

2.4.3 Summary of current ecological understanding of population dynamics

The exact mechanisms determining abundance are difficult to untangle in complex natural environments. However, what is evident from the above discussion is that habitat availability and quality do influence population abundance whether actively by stimulating competition and predation or passively, for example, through starvation. Habitat availability and heterogeneity controls carrying capacity and also the environment's resilience or sensitivity to both anthropogenic and natural pressures. In terms of developing the DPSIR framework changes to the 'State' of salmonid habitat can 'Impact' salmonid abundance by a number of mechanisms including, competition, predation, starvation, disease, asphyxiation, and injury (either physical or chemical). The ability of an individual fish to survive in an environment or withstand pressures is potentially influenced by their level of mobility. It is also worth remembering that this may also be influenced by the individual fitness of that fish and therefore by its genetic predisposition (Milner *et al.*, 2003).

Restoration strategies that either: (1) increase carrying capacity, for example, by increasing food supplies, refugia or habitat heterogeneity; (2) reduce the ability for negative impacts to operate, for example, increasing cover to reduce competition by increasing visual isolation and reducing interaction between fish (Summers *et al.*, 1996); or (3) increase environmental resilience such as riparian buffer zones reducing diffuse pollution, should therefore result in increased abundance. However, the success of such restoration strategies relies on the ability to identify which habitat component is critical in limiting carrying capacity. This requires knowledge of habitat requirements at different life-stages and is the next focus of this review.

2.5 Current ecological understanding of habitat requirements

Salmonid in-stream habitat represents the 'State' component of the DSPIR framework. It is essential to identify which components of the in-stream environment are ecologically relevant to salmonids so that these in turn can be used to identify which hydrological and geomorphological processes are most likely to result in ecologically relevant impacts. It is also important to identify the optimum 'State' of these components for salmonids so that areas of degradation can be located and restoration strategies identified aimed at reinstating or creating optimum habitats.

In Chapter One, Table1.1 and Figure 1.2 presented a summary of the Atlantic salmon and brown trout life-cycle, together with a board overview of habitat requirements. The aim of this section is to expand upon this summary for the freshwater stages of the life-cycle principally spawning, fry and parr habitat requirements, to identify critical and ecologically relevant components of the in-stream environment, or alternatively, critical salmonid 'resources', that if degraded may result in one or more of the 'Impacts' (e.g. competition, predation, starvation, disease, asphyxiation, injury, or toxicity) discussed above. It is important to recognise that whilst the habitat requirements of salmon and trout do overlap there are several important differences between the two species as discussed in the following sections.

2.5.1 Salmonid spawning habitat requirements

In the Northern Hemisphere, Atlantic salmon and brown trout have been observed spawning between October and March, with the peak time being in November and December (Elliott, 1994; Summers *et al.*, 1996; Armstrong *et al.*, 2003). At this time, breeding fish migrate upstream, typically to their own natal stream to find suitable spawning grounds. Whilst some fish, primarily Atlantic salmon, may spawn in main stem rivers, spawning grounds, particularly for trout, are often located within small low-order tributaries (Armstrong *et al.*, 2003). Initiation of spawning activity usually follows a period of high flows that enable adult fish to enter these smaller tributaries (Soulsby *et al.*, 2001). The first habitat requirement is therefore access to spawning grounds (e.g. Crisp, 1996). Any obstruction whether physical or chemical, anthropogenic or natural, temporary or permanent that disrupts the longitudinal in-stream connectivity of migratory runs, fragmenting adult and spawning habitat may be detrimental to spawning success. Inability to reach spawning grounds may result in: (1) reduced numbers of fish spawning; (2) fish spawning in unsuitable or less suitable habitats, thereby reducing survival to emergence; or (3) increased and more concentrated spawning below obstructions. In the latter situation this could lead to competition for space which at very high densities may cause overcutting of redds with mortality of eggs (Solomon, 1985 cited from Milner *et al.*, 2003). Increased spawning densities may also lead to knock-on effects for the juvenile life-stage with increased competition and higher mortality rates. Factors ('Pressures') that may cause obstruction include, physical barriers such as dams weirs, rapids and waterfalls (e.g. Crisp, 1996; Calles and Greenberg, 2005) or chemical factors such as plugs of deoxygenated or toxic water in the estuaries or lower reaches of rivers (Crisp, 1996; Solomon and Sambrook, 2004). Additionally, high temperatures or low flows may hamper spawning by failing to initiate migratory activity (Solomon and Sambrook, 2004).

On reaching spawning grounds, fish lay and fertilise eggs within a redd (nest) excavated from gravel substrate (Figure 2.4). Redd site selection and excavation is generally undertaken by the female fish who 'cuts' a depression into the gravel by vigorously flexing her body and tail to generate up-currents, dislodging gravel, causing it to be displaced downstream. Once satisfied that the depression is complete, the female lays her eggs which are simultaneously fertilised by a male fish. The fertilised eggs exhibit negative buoyancy which causes them to sink and settle within the gravel interstitial spaces. The female then moves slightly upstream repeating the 'cutting' procedure to create a new depression whilst burying the eggs in the downstream depression with the newly displaced gravel. This procedure is repeated until all her eggs have been laid (Elliott, 1994; Summers *et al.*, 1996, Soulsby *et al.*, 2001). Atlantic salmon typically bury their eggs at a depth of 13-30cm whilst brown trout cut shallower depressions of 8-25cm (Crisp, 1989). This is likely to represent a correlation with fish size and therefore their ability to move substrate (Crisp, 1996).

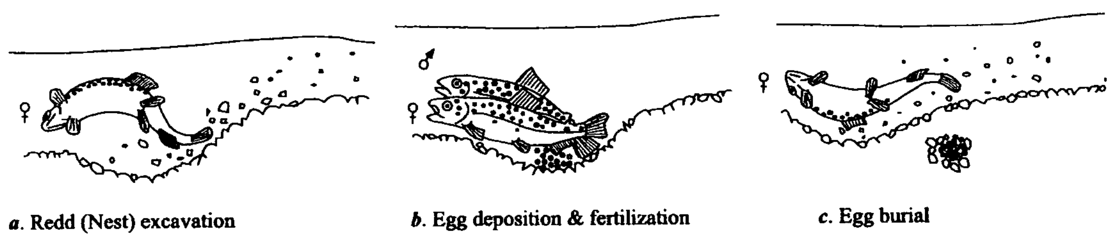


Figure 2.4: *Salmonid spawning behaviour: (a) redd excavation; (b) egg deposition; and (c) egg burial. Reproduced from Soulsby et al., (2001).*

Redd site selection and excavation relies in part on the second set of spawning habitat requirements considered here, which are the physical habitat conditions of velocity, depth and substrate. These factors control an individual fish's ability to dislodge gravel, to hold position (station) within the water column whilst redd building and for dislodged sediment to be displaced downstream. They also determine the structure of the redd and the ability for throughflow to ventilate the redd and provide a constant supply of oxygen to the eggs (Armstrong *et al.*, 2003). Numerous studies have been undertaken on this aspect aimed at identifying the optimum hydraulic and substrate conditions for spawning and a number of comprehensive review papers have been produced synthesising this research (e.g. Crisp, 1996; Summers *et al.*, 1996; Hendry and Cragg-Hine, 1997; Armstrong *et al.*, 2003). A summary of the review paper findings are presented in Tables 2.4(a&b) showing that velocity, depth and substrate preferences are highly variable both between and within species.

Table 2.4 (a) *Observed spawning habitats utilised by brown trout (based on Summers et al., (1996) and Armstrong et al., (2003).*

Water Velocity ($cm\ s^{-1}$)			Water Depth (cm)			Substrate size (mm)			References
Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	
15	20							15	Crisp and Carling (1989)
15	75	39.4	6	82	31.7	8	128	14	Shirvell and Dungey (1983)
10.8	80.2	46.7			25.5			6.9	Witzel and MacCrimmon (1983)
20	60		20	50					Johnson *et al.*, (1995)
35	95		18	46					Nelson (1986)
24	37		12	18		26	75		Grost *et al.*, (1990)
30	40					8	128	80	Ottaway *et al.*, (1981)
20	70		20			0.75	7.5		Raleigh *et al.*, (1986)
						8	128		Chapman (1988)

Table 2.4 (b) *Observed spawning habitats utilised by Atlantic salmon (based on Hendry and Cragg-Hine, (1997), and Armstrong et al., (2003).*

Water Velocity ($cm\ s^{-1}$)			Water Depth (cm)			Substrate size (mm)			References
Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	
20									Crisp and Carling (1989)
		40			50			100	Heggberget (1991)
		53			25			20.7	Moir *et al.*, (1998)
35	80	53	17	76	38				Beland *et al.*, (1982)
25	90		25						Reiser (1991)
36	76		20	30					Petersen (1978)
								112.8	Ottaway *et al.*, (1981)

What is less clear, due to the high degree of correlation between these three variables, is which variable is the dominant cue for spawning site selection (Crisp, 1996; Armstrong *et al.*, 2003). It has been suggested that velocity may be the dominant variable with fish having a preference for higher velocities that enable greater flow through the redd and hence higher rates of oxygen delivery to the incubating eggs. Female fish have been observed 'testing' velocities within redds by lowering their anal fin into the depression (Crisp, 1993, *cited in* Armstrong, *et al.*, 2003). However, it has also been suggested that velocity may simply be a proxy for substrate size which ultimately determines redd structure (Shirvell and Dungey, 1983 *cited in* Armstrong *et al.*, 2003). Irrespective of which is dominant, it is likely that the upper constraints on velocity and substrate selection may in part be related to variations in fish size. Larger fish are able to hold station at greater velocities and to dislodge coarser substrate particles (Summers *et al.*, 1996). In terms of depth, larger female fish require deeper water than smaller fish, as they are unlikely to spawn in water much shallower than their own body depth (Armstrong *et al.*, 2003). In this respect salmon, which are typically larger than trout, may utilise main stem rivers and larger tributaries for spawning, whilst trout utilise the smaller headwaters. The high variability in velocity, depth and substrate preferences observed between and within the two species, may therefore be related to a high variability in fish size. This variability has led some researchers to use dimensionless variables such as the mean Froude number (Soulsby *et al.*, 2001) to characterise habitat selection, as these standardise for different sized rivers allowing comparison between sites.

The availability of suitable spawning habitat in terms of substrate, velocity and depth can have a major control upon productivity. Anthropogenic pressures that reduce the availability of suitable spawning habitat include: (1) canalisation or dredging of channels resulting in flow homogenisation and the creation of typically deep slow flowing channels which are unsuitable for spawning; (2) gravel extraction may remove valuable spawning gravels from the river; (3) bank protection measures aimed at protecting property or agricultural land from erosion may also reduce gravel additions to the channel by reducing natural bank erosion; (4) obstructions such as dams and weirs may disrupt downstream sediment transport preventing gravel replenishment over time (Kondolf, 1997); and (5) flow regulation may result in fish spawning in areas of gravel at times of high flow which then become exposed during low flows resulting in mortality through freezing or desiccation (Crisp, 1996).

Following spawning, the third critical requirement for survival and embryonic development is a good and constant supply of well oxygenated water to the eggs (Crisp, 1996; Armstrong *et al.*, 2003). One habitat factor which can severely impact upon this is the infiltration of fine sediment (fines) into the redd. For example, egg mortalities of up to 86% were recorded in one stream where redd siltation occurred (Soulsby *et al.*, 2001). A number of studies have been undertaken investigating the effect of fine sediment infiltration upon salmonid spawning and survival to emergence noting impacts such as: (1) reduced flow through the redd reducing oxygen delivery to eggs (Greig *et al.*, 2005) and impeding removal of metabolic waste products such as ammonia (Crisp, 1996); (2) reduced oxygen concentrations within the interstitial water, which may be particularly significant where organic matter infiltrates the redd and decomposition reduces oxygen tensions (Soulsby *et al.*, 2001); (3) infiltration of clays, even in small quantities may restrict the exchange of oxygen across the egg membrane through the development of a sedimentary seal around individual eggs (Greig *et al.*, 2005); (4) nutrients such as phosphorus and toxins including pesticides and herbicides attached to fine sediment particles may also infiltrate the redd causing mortality and reduced growth (Lower and Moore, 2003); and (5) the presence of fines can reduce the ability of alevins to wriggle out through gravels and emerge from the redd (Summers *et al.*, 1996). The process of redd cutting is considered to clean gravels as fine sediments are disturbed and carried downstream increasing porosity within the redd (Summers *et al.*, 1996; Armstrong *et al.*, 2003; Greig *et al.*, 2005). However, whilst this may be the case, researchers have also observed rapid re-infiltration of fines into redds by the time of emergence. For example, Soulsby *et al.* (2001) observed complete siltation of artificial redds within a week, and possibly within a single moderate to large storm event in a Scottish lowland tributary. Sources of fine sediment and 'Pressures' which may increase fine sediment loads and hence are hypothesised to increase the risk of redd siltation include: (1) runoff from agricultural fields both arable and stocked; (2) channel bank erosion; and (3) road and urban runoff.

In order to enhance the flow of oxygen and water through the redd it has been proposed that salmonids tend to spawn at the upstream end of riffles. At this point, velocities are increasing and the difference in hydraulic head between the upstream pool and downstream riffle causes water to be drawn down into the streambed (the hyporheic zone) (Summers *et al.*, 1996; Petts, 2000). In upland environments that lack pool-riffle sequences a similar effect has been observed around protruding boulders (Crisp, 1996). A number of researches have suggested that the source of water (groundwater or surface) that dominates within the hyporheic zone is vitally important for egg survival. Groundwater of long residence time typically has a higher mineral content and lower oxygen content than surface water. Therefore, in areas of groundwater upwelling, egg mortality may be high due to a reduced oxygen supply (Soulsby *et al.*, 2001). However, others suggest that areas of upwelling groundwater may be favourable, due to higher water temperatures during winter that prevent freezing and accelerate growth, whilst upwelling may also help to limit fine infiltration (Summers *et al.*, 1996; Malcolm *et al.*, 2002). The selection and effect of upwelling or downwelling water appears highly variable between locations and may depend upon local conditions.

The final habitat requirement discussed here that does not appear to have been studied to any great extent is that of cover for adult fish whilst spawning. Several researchers have suggested the need for cover in the form of deep pools, boulders, tree roots, and overhung banks in close proximity to spawning grounds to provide protection from bright sunlight and predation whilst resting as redd building can take several days (Crisp, 1996; Armstrong *et al.*, 2003).

2.5.2 Juvenile fry and parr habitat requirements

Upon hatching the young fish are known as alevins which remain in the gravel, feeding off their yolk sac. Just before this is exhausted, they wriggle up through the gravel and swallow an air bubble to attain neutral buoyancy (Crisp, 1996). Following absorption of the yolk sac juvenile salmonids begin actively feeding and become known as fry. There is some debate within the scientific literature as to how long this stage (terminology) persists (Summers *et al.*, 1996). The term 'fry' has been used to describe juveniles: (1) only until dispersal from the redd; (2) up until the end of their first summer; and (3) for the whole of their first year of life. For the purposes of this thesis, juveniles will be referred to as fry utilising nursery habitat until the end of their first summer and as parr utilising rearing habitat thereafter. Fish generally remain as parr for 1 to 3 years after which time they either undergo smoltification and migrate to sea (Atlantic salmon and sea trout) or become sexually mature as adult brown trout, remaining in their natal river or migrating to main stem rivers, lakes or estuaries (Elliott, 1994).

2.5.2.1 Physical habitat requirements

As for spawning, numerous studies have been undertaken aimed at describing the preferred physical habitat use of fry (Tables 2.5 a-d). The fry period is considered to be critical with high mortality as a result of limited dispersal, resulting in high densities and therefore high levels of competition for food and territories (Elliott, 1994; Summers *et al.*, 1996; Armstrong *et al.*, 2003). The proximity of fry habitat to spawning habitat is therefore crucial in determining survival. In general, fry habitat for both species appears to be found within areas of shallow, fast flowing water, typically riffles, of moderately coarse pebble to cobble substrate (Hendry and Cragg-Hine, 1997). These areas provide a number of advantages for salmonid fry: (1) surface turbulence provides cover from predation (Hendry and Cragg-Hine, 1997); (2) turbulent flow increases the dissolved oxygen content of water which is particularly important for fry as oxygen consumption is inversely related to per unit body weight (Crisp, 1996); (3) riffles are areas of high food production, light penetration through the water column is inversely related to water depth, therefore at shallower depths primary production is maximised resulting in a good supply of drift invertebrates to fish; (4) the coarse pebble to cobble substrate provides shelter and cover that is adequate for fry but free from competition and predation by larger fish which require coarser substrate (Armstrong *et al.*, 2003); and similarly (5) water depths are adequate for fry but unsuitable for larger fish enabling them to avoid competition (Summers *et al.*, 1996).

Within this broad habitat classification there is evidence and discussion in the scientific literature for further microhabitat selection. Fry are not yet powerful swimmers, and to minimise energy expenditure, occupy relatively low velocity zones close to the channel bed (Summers *et al.*, 1996). However, at the same time, higher velocities are required to deliver food. The most efficient strategy for maximising net energy gain therefore appears to be to hold position within a low velocity flow, but one that is adjacent to a high velocity current supplying drift invertebrates at high rate (Armstrong *et al.*, 2003). Different metrics such as vorticity and circulation are being developed and tested to capture these subtle velocity arrangements and apply them within habitat studies (e.g. Crowder and Diplas, 2002).

As fish grow into parr they become capable of utilising a much greater range of velocities (Summers *et al.*, 1996) and also disperse towards deeper habitats (Table 2.5), which may be faster flowing zones such as runs and rapids with coarser cobble, boulder substrate, slower flowing, silty pools and glides or even lakes (Armstrong *et al.*, 2003; Crisp, 1996). Deeper water and/or coarser substrate are required to meet the increased cover demands of larger fish, whilst feeding within higher velocities enables a greater intake of food to meet their increased energy demands. At this stage, they are more able to adapt to a variety of conditions and therefore to utilise the habitat available to them even if it does not meet optimum conditions. At both the fry and parr life-stage juvenile salmon are better adapted to higher velocities than juvenile trout of the same size, as a result of their large pectoral fins which act as hydrofoils and aid them to maintain position (Crisp, 1996). Therefore, salmon fry are typically found in shallower, faster flowing zones, whilst trout fry exhibit a preference for deeper areas of slower velocity, often located at riffle margins. Trout are considered the more aggressive and territorial species and in areas of coexistence juvenile trout have been observed to out-compete salmon fry for low velocity areas (Heggenes *et al.*, 1999).

Table 2.5 (a) *Observed fry habitats utilised by brown trout (based on Summers et al., (1996) and Armstrong et al., (2003).*

Water Velocity ($cm\ s^{-1}$)			Water Depth (cm)			Substrate size (mm)			References
Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	
0	30		15	30					Belaud *et al.*, (1989)
		41			38				Bird *et al.*, (1995)
0	40			60					Bovee (1978)
			10	20		50	70		Heggenes (1988)
				20					Kennedy and Strange (1982)
0	20			40					Loar (1985)
			10						Larsen (1972)
	30			60					Lambert and Hanson (1989)
6	30		30	60					Raleigh *et al.*, (1986)
0	20		20	30		10	90		Bardonnet and Heland (1994)
			5	35					Maki-Petays *et al.*, (1997)

Table 2.5 (b) *Observed fry habitats utilised by Atlantic salmon (based on Hendry and Cragg-Hine, (1997), and Armstrong et al., (2003).*

Water Velocity ($cm\ s^{-1}$)			Water Depth (cm)			Substrate size (mm)			References
Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	
20	40								Crisp (1993)
15	100			10					Heggenes *et al.*, (1999)
10	30								DeGraaf and Bain (1986)
			20	40					Morantz *et al.*, (1987)
20	75			25		16	64		Symons and Heland (1978)
35	40		40						Bagliniere & Champigneulle (1986)
39.3	57.0		21.7	23.8		90	143		Borgstrom (1991)

Table 2.5 (c) *Observed parr habitats utilised by brown trout (based on Summers et al., (1996) and Armstrong et al., (2003).*

Water Velocity (cm s^{-1})			Water Depth (cm)			Substrate size (mm)			References
Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	
0	40				30				Belaud *et al.*, (1989)
		38			42				Bird *et al.*, (1995)
0	50	5		90					Bovee (1978)
5	30		30	60					Heggenes and Saltveit (1990)
15	60		22	55					Johnson *et al.*, (1995)
0	30				30				Loar (1985)
0	50				30				Moyle *et al.*, (1983)
9	45		27	57					Shuler *et al.*, (1994)
10	70				>50		128		Heggenes (1988)
0	65	26.7	14	122	65				Shirvell and Dungey (1983)
			40	75					Maki-Petays *et al.*, (1997)
						8	128		Eklov *et al.*, (1999)

Table 2.5 (d) *Observed parr habitats utilised by Atlantic salmon (based on Hendry and Cragg-Hine, (1997), and Armstrong et al., (2003).*

Water Velocity (cm s^{-1})			Water Depth (cm)			Substrate size (mm)			References
Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	
20	60		20	70		64	512		Heggenes *et al.*, (1999)
	120		25	60					Morantz *et al.*, (1987)
50	65		25	60		64	512		Symons and Heland (1978)
0	60	28							Shustov (1990)

2.5.2.2 The requirement for cover

For both fry and parr, habitats with a large amount of cover are considered beneficial (Summers *et al.*, 1996; Hendry and Cragg-Hine, 1997; Heggenes *et al.*, 1999; Armstrong *et al.*, 2003; Johnsson *et al.*, 2004; Johansen *et al.*, 2005). Cover can be provided by bankside features such as undercut banks, exposed tree roots, and shade from overhanging vegetation, or in-stream features such as boulders, woody debris, and submerged macrophytes (Summers *et al.*, 1996). Deep pools (Elso and Giller, 2001) or turbulent water (Armstrong *et al.*, 2003) can also act as a form of cover in reaches lacking the aforementioned features. The presence of cover is understood to provide a number of benefits.

First, cover provides protection from predation. Experiments have found that fish defend territories with cover more aggressively and to a greater extent than those without, especially following an increase in the perceived threat from predation (Johnsson *et al.*, 2004). Cover from predation is thought to be particularly important during the winter and the availability of over-wintering habitat for salmonid parr has received considerable attention in the recent scientific literature (Valdimarsson and Metcalfe, 1998; Armstrong and Griffiths, 2001; Armstrong *et al.*, 2003; Riley *et al.*, 2006;). In general, as temperatures drop below about 10^{o}C, the ability for salmonids to hold station within high velocity currents diminishes. At such temperatures, fish have been observed to switch to nocturnal activity, sheltering within substrate interstices (typically coarser than those used in summer), macrophytes, and tree roots by day and moving to feeding stations at night. A main benefit of sheltering is thought to be the avoidance of predators which, if endothermic, may enjoy an increasing advantage in speed over the ectothermic fishes as water temperature declines (Valdimarsson & Metcalfe, 1998);

Second, it may provide increased visual isolation. This reduces territorial behaviour and competition between fish, and hence increases productivity. This is thought to be particularly important for juvenile trout, which are especially territorial (Heggenes *et al.*, 1999). Territorial competition is considered to be central to the mechanisms responsible for population regulation in brown trout (Elliott, 1994) and observations have concluded that a dominant trout will not tolerate any other fish within its visual range displacing subordinate fish by aggression (Summers *et al.*, 1996).

Third, overhead cover provides shade which may have impacts upon territory size and competition. Experiments have shown the level of aggression and competition between salmon to increase with light intensity (Valdimarsson and Metcalfe, 2001). Therefore, in shady reaches, fish may occupy smaller territories resulting in higher densities. Additionally, shade may be important during periods of high summer temperatures, and may become even more important should the predicted rises in temperature as a result of climate change occur. Temperatures over 20°C for salmon and 19°C for trout are believed to be detrimental (Crisp, 1996).

Fourth, it may generate flow complexity. In-stream features such as woody debris and boulders are associated with the generation of complex flow patterns. This may enhance the development of low velocity zones in close proximity to high velocity zones, thereby increasing the density of profitable feeding sites and hence fish. However, this is not without debate. Much of the research regarding the benefits of woody debris has been undertaken in the Pacific North West, with respect to Pacific pool dwelling salmonids. There are arguments that it is less important to Atlantic salmon that are predominantly riffle dwelling (Summers, 2000). It has also been suggested that too much in-stream vegetation may reduce flow velocities to such an extent that fine sediment deposition occurs limiting invertebrate production and hence reducing salmonid productivity (Summers *et al.*, 1996).

Finally, in-stream cover and encroaching bankside vegetation may provide important refugia for fish during periods of extreme flow both floods and droughts.

2.5.2.3 Food sources and feeding habits

In respect of food sources and feeding habits an important distinction between the two species should be made. Juvenile salmon are considered to be reliant on autochthonous production and aquatic, macroinvertebrates for food (e.g. mayfly, stonefly and caddis larvae), much more so than the more opportunistic juvenile trout that will also feed on terrestrial invertebrates which fall into the channel. To revisit the DPSIR model and the structure of the 'State' category, these feeding (biotic) requirements can in turn be related to hydrological and abiotic requirements. First, the widely cited research by O'Grady (1993), found that in reaches of tunnelled (heavily shaded) vegetation, juvenile salmon and trout stocks were only 19.4% and 28.5% respectively of those found in open reaches. This result has been primarily attributed to a decline in food availability due to the impact of reduced light penetration on autochthonous primary productivity and aquatic invertebrate abundance in tunnelled reaches (Summers, 2002). Taking feeding habits into account the impact of excessive shade is assumed to be more detrimental for juvenile salmon than juvenile trout. Conversely, overhanging vegetation may actually be an important food source for trout as it encourages terrestrial invertebrates to fall into the channel. The amount of riparian vegetation has been shown to explain up to 93% of the variation in the drift density of terrestrial invertebrates (Johansen *et al.*, 2005), which have been found to contribute up to 91% of trout prey (Kelly-Quinn and Bracken, 1990 *cited in* Johansen *et al.*, 2005). Some overhanging vegetation is also likely to be important for salmon as it is a vital habitat component for the adult reproductive stage of many aquatic insect life-cycles (Hendry and Cragg-Hine, 1997). In-stream vegetation and macrophytes such as *Ranunculus* species are also likely to be important as they support aquatic invertebrates at the larval stage (O'Connor and Kennedy, 2002). However, as Armstrong *et al.* (2003) note, there appears to be no data available to predict exactly how much vegetation or cover is optimal in different catchments.

Second, autochthonous production also requires the delivery of nutrients to the channel. Catchment nutrient sources can be associated with land use and geology, and their availability may be determined by the ease with which they are transported to the channel by a combination of surface, sub-surface and groundwater flow pathways (e.g. Croke and Hairsine, 2006). Bankside vegetation may also increase nutrient availability through the provision of leaf litter (Armstrong *et al.*, 2003). In reaches of low nutrient status (oligotrophic waters) autochthonous production may be limited resulting in either a lower abundance of salmonids and/or smaller fish as a result of slower growth rates (Folt *et al.*, 1998). In reaches of high nutrient status (eutrophic waters) excessive in-stream production may result in a high biological oxygen demand and again reduced numbers of fish. As with cover there appears to be little data available to predict exactly what level of nutrients is optimal in different catchments.

It terms of the impact of changing food resource availability there again appears to be no clear answer within the scientific literature as to the impact on salmonid production (Folt *et al.*, 1998). For example, does increasing food result in more fish, larger fish or more, larger fish? It has also been suggested that growth rates may be linked to the tendency for salmon to: (1) migrate to sea at a younger age; (2) for females to return as grilse or multi-sea-winter fish; and (3) affect the number of eggs produced (Armstrong *et al.*, 1998). These characteristics are primarily considered to be genetically determined, but if environmental influences in food availability are also important, then changing light or nutrient and hence food availability may have considerable consequences for fisheries managers (Armstrong *et al.*, 1998).

2.5.3 Water Quality

Salmonids are often cited as an excellent indicator of freshwater ecosystem health due to their requirements for clean high quality water. The issue of siltation has already been raised in conjunction with spawning, but excessive inputs of fine sediment can also be detrimental to fry and parr by reducing aquatic invertebrate densities, causing damage to gills and reducing visibility hindering foraging (Watts *et al.*, 2003). Salmonids also require water with a high dissolved oxygen content at all stages of their life-cycle. The precise levels required vary with activity, feeding and temperature, but levels from 5 mg l^{-1} to 9 mg l^{-1} and 80% saturation have been suggested within the literature (Crisp, 1996; Armstrong *et al.*, 2003). Eutrophication of waters and increased biological oxygen demand (BOD) have therefore, been suggested as significant factors in the decline of salmonid populations. Whilst naturally nutrient-rich water may be beneficial in aiding in-stream production, providing a plentiful supply of invertebrates, artificially high nutrient loadings can result in excessive primary production with the development of significant algal blooms and increased BOD. Such issues are particularly evident during summer at times of low flow and high temperatures. A range of other toxic chemicals, that have been found to adversely effect salmonid survival, can also been found in freshwaters including, pesticides, herbicides, endocrine disruptors, heavy metals, surfactants, microbial pathogens, salts and hydrocarbons (e.g. Lower and Moore, 2003; Heneay *et al.*, 2001; Waring and Moore, 2004; Oliver *et al.*, 2005, Sanzo and Hecnar, 2006) . Salmonids are also sensitive to extremes of pH, and there is evidence to suggest that pH greater than 9.0 or less than 4.5 may be detrimental. As well as direct effects, pH together with temperature can affect the toxicity of many other chemicals, one of the most widely cited, being aluminium (Crisp, 1996).

2.5.4 Differences between salmon and trout

The habitat requirements of salmon and trout at the spawning, fry and parr life-stages do exhibit a great deal of overlap. However, there are also subtle differences between the two species leading to a number of papers discussing the importance of niche separation in enabling them to coexist (e.g. Heggenes *et al.*, 1999; Riley *et al.*, 2006). Greater differences in habitat use have been observed between salmon and trout and for different age classes than between summer and winter (Riley *et al.*, 2006). These have been attributed to anatomical, physiological, and behavioural differences which result in different hydraulic, abiotic and biotic requirements (Table 2.6)

Table 2.6: *Differences between salmon and trout at the spawning, fry and parr life-stages.*

State component	Juvenile salmon	Juvenile trout
Hydraulic	Large pectoral fins enable juvenile salmon to hold station at higher water velocity and to dominate faster flowing zones of shallower water. As adults, salmon are larger and therefore, more able to utilise main stem rivers and larger tributaries for spawning.	Less able to hold station at high velocities trout exhibit a preference for deeper areas of slower velocity, often located at riffle margins, for which they will out-compete salmon. In general, as adults, trout are smaller and therefore, more able to utilise smaller headwaters for spawning
Abiotic	Lower levels of aggression and territoriality mean overhead cover is less important for salmon.	Cover is important for trout to reduce intra-specific competition.
Biotic	Reliant on autochthonous production and aquatic invertebrates. Light and nutrient availability are important.	Terrestrial invertebrates can contribute a significant proportion to trout prey. This can be associated with the amount of riparian vegetation.

These considerations have resulted in the conclusion that juvenile trout may perform well in narrow streams where bankside shelter is abundant relative to the area of the stream bed, whilst salmon which are less dependent upon cover may thrive in wider streams where the bankside has less influence and where they are free from competition with trout (Armstrong *et al.*, 2003). Indeed there are a number of references to trout, in particular, making use of very small tributaries, for example, less than 0.8m wide in Black Brows Beck, Cumbria (Elliott, 1994), and less than 2.75m in tributaries of the River Usk (Bembo *et al.*, 1993). This presents an interesting situation in that different species utilise different scales of habitat to differing degrees, according to their specific habitat requirements. These different habitat scales are found in different locations of the catchment. Therefore, it may be that restoration strategies aimed at dealing with particular pressures and habitat controls may be applicable at different habitat scales in different locations. For example, riparian controls that influence cover availability, such as intensive grazing and agricultural stock access, may be more effective in smaller trout producing streams than in wider salmon producing rivers. These issues are yet to be explored experimentally (Armstrong *et al.*, 2003) particularly at a catchment-scale leading to the development of Hypothesis (2).

Hypothesis 2: Relationships between habitat and salmonid abundance are species and location specific relating to the scale of habitat occupied by different species.

2.5.5 Summary of current understanding of salmonid habitat requirements

What is clear from the above discussion is that salmonids utilise a complex mosaic of habitats, both spatially and temporally, for spawning, feeding, sheltering, and as refuges during winter, floods and droughts. The more complex (heterogeneous) the environment, the greater the carrying capacity and probability that fish will be able to find the habitat they require within their range, free from competition. It is especially crucial that fry habitat be located close to spawning sites due to their limited dispersal capabilities and therefore, the spatial arrangement of habitat is important. As discussed in Section 2.4.2, the more homogeneous the environment, the further fish have to move to find the habitats they require; such movements are associated with greater predation risk and utilise more energy. As fish grow they are more able to adapt to their surroundings and make greater movements. However, at the same time, they are also likely to need a wider variety of habitats. Pressures that may reduce in-stream heterogeneity include: (1) intensive grazing within the riparian zone, reducing vegetation cover and causing the collapse of undercut banks; (2) deforestation and the removal of bank side trees; (3) urban development within the riparian zone including bank reinforcement; and (4) canalisation, dredging, gravel extraction, weed cutting and debris clearance for flood conveyance and navigation purposes. The availability of food is also important and this may be related to light and nutrient availability as controlled by geology, land use, flow pathways and riparian tree cover. Salmonids also demand their environment to be supported by water of high quality, which due to their complex life-cycle and extensive range is required across large areas of the catchment. This confirms the approach outlined in Chapter One which stated the need for a catchment-wide analysis of salmonids and their habitat that is spatially explicit. Pressures that may reduce water quality include: (1) runoff from agricultural land; (2) discharges from sewage treatment works and septic tanks; (3) industrial effluents; (4) road runoff; (5) mine drainage; and (6) runoff from forestry. The precise requirements of salmonids have been found to be species-specific and are therefore likely to be location specific as the different species utilise different scales of habitat in different locations throughout their life-cycle. The next sections of this review concentrates on the geomorphological and hydrological controls, together with anthropogenic pressures, which influence the distribution and abundance of habitat heterogeneity, cover availability, food availability and water quality.

2.6 Current understanding of geomorphological and hydrological controls on in-stream conditions

It is increasingly recognised that aquatic habitats and hence ecology are intrinsically linked to geomorphological and hydrological processes (Newson and Newson, 2000; Piegay *et al.*, 2000; Fukushima, 2001; Gilvear *et al.*, 2002; Walters *et al.*, 2003). Indeed many models aimed at predicting in-stream ecological diversity and abundance give considerable weighting to in-stream geomorphological and hydrological variables (Gilvear, 1999), e.g. HABSCORE (Milner *et al.*, 1993), for predicting fish carrying capacity, SERCON (Boon *et al.*, 1998), for evaluating river conservation status and RIVPACS (Wright *et al.*, 1998) for predicting invertebrate assemblages. In the previous section, the requirements of salmonids were discussed making reference to preferred hydraulic and substrate conditions, whilst also raising the importance of specific morphological features such as riffle-pool sequences, as locations where preferred conditions are most likely to occur. The spatial distribution of these conditions and features has in turn been associated with geomorphological and hydrological variables such as channel slope, channel planform, width:depth ratios and flow regime. For example, riffle-pool sequences typically correlate with channel slopes < 2% whilst channel slopes >4% exhibit step-pool sequences (Sear *et al.*, 2003). In a study relating fish assemblages to geomorphological variables Walters *et al.* (2003) concluded that local stream slope was the dominant factor controlling stream habitat. Fukushima (2001), observed a correlation between redd density and channel sinuosity in low gradient alluvial streams, commenting that sinuosity is a primary control over the density of riffle-pool sequences in such environments. However, it should be recognised that these variables may not be the ultimate drivers of in-stream habitat but rather effective descriptors or 'surrogate' variables for the overriding controls of geology, topography, climate, land cover and flow regime. It is important to recognise these natural controls over fish abundance and distribution, as restoration may not be applicable everywhere within a catchment. Some locations may simply be unsuitable for salmonids.

The disciplines of fluvial geomorphology and hydrology aim to understand the processes controlling the spatial distribution of these variables and as such, are primarily focused upon understanding energy, sediment and water transfers through the fluvial system. In achieving this, the concepts of connectivity and coupling are central, as paralleled in ecology by definitions of connectivity describing the movement of species throughout landscapes (Moilanen and Hanski, 2001). These concepts have been widely applied to the study of many phenomenon within the disciplines of hydrology and fluvial geomorphology including: landscape sensitivity (Brunsden and Thornes, 1979); buffer zone function (Haycock and Burt, 1993); the flood pulse concept (Middleton, 1999); the impact of mass movements upon sediment delivery (Harvey, 2002); and more recently the impact of landscape characteristics such as land use and topography upon runoff generation, the delivery of sediment and contaminants and flood production (Burt, 2001; Lane *et al.*, 2003a; Lane *et al.*, 2006; Bracken and Croke, in press).

In terms of conceptualising river systems and the processes operating within them, Schumm (1977) developed the theory of a three zone longitudinal continuum with an upstream erosion zone, connected to a transition zone of sediment transport, ending with a downstream deposition zone. From this the concept of downstream trends in many fluvial properties such as sediment fining (Knighton, 1980), river width:depth ratios, and longitudinal profile (channel slope) have been developed. Similar comparisons of continuum theory can be found within ecological research with the River Continuum Concept (RCC) (Vannote *et al.*, 1980) that describes downstream changes in species diversity and abundance driven by downstream gradients in nutrient and energy levels. Both these theories can be related to the premise as described by Hynes, (1975), "that in every respect the valley rules the stream" (*cited in* Johnson and Gage, 1997). In other words, it is landscape variables such as topography, climate, geology, and soil type which ultimately control the distribution of these processes and hence the distribution of habitat. However, as Piegay *et al.* (2000) note, while these continua may manifest at the catchment scale, channel form and habitat at the local and reach scale are typically not well predicted, but are instead influenced by local discontinuities in geomorphological and hydrological processes. For example, disruptions to downstream trends in bed elevation, channel gradient and bed material size are often observed downstream of lateral water and sediment inputs such as at tributary junctions or landslides (e.g. Ferguson *et al.*, 2006). Discontinuities can also occur due to localised changes in geology, glacial legacy, land use and anthropogenic actions. Similarly, within ecology the River Continuum Concept has been acknowledged as an effective framework for catchments but that within individual reaches, longitudinal relationships may be obscured by local factors resulting in the detailed adaptation of biota to ecosystem niches (Newson and Newson, 2000). This has resulted in the promotion of patch dynamic theory, and concepts such as physical biotopes (Padmore, 1998), to characterise patterns and processes in heterogeneous environments. However, difficulties have arisen because the processes influencing the spatial and temporal distribution of hydraulic variables and ecological patches are often poorly understood (Walters *et al.*, 2003).

What is now more widely recognised is that the local and catchment perspective are intrinsically linked and that large scale processes create the template within which the small scale operates (Armstrong *et al.*, 1998; Stauffer *et al.*, 2000). This poses an interesting challenge when investigating environmental systems as discussed in Chapter One, in that different scales of analysis may identify different scales of process as important in controlling environmental conditions. This precise issue was recognised by Wiley *et al.* (1997) who argued that viewing the same ecological system from both a site and landscape perspective can lead to contradictory interpretations as to the controlling factors. At a large spatial scale ecological communities appear predictable and related to large scale processes such as catchment geology and hydrological regime (as in continuum theories), but as the spatial scale contracts high variability is observed and local factors become emphasised more (as in patch dynamics). This, Allan and Johnson (1997) state, leads to a hierarchical organisation of physical units most clearly captured for rivers in the hierarchy habitat-reach-segment-subcatchment-basin. Concepts adopting this hierarchical approach are emerging within the literature. For example, Thorp *et al.* (2006) propose the "riverine ecosystem synthesis". They portray rivers as a downstream array of large hydrogeomorphic patches formed by catchment geomorphology and climate within which "functional process zones" are formed according to physiochemical habitat differences. Community structure and ecosystem function are then theorised to vary predictably among different types of functional process zone. However, to date few field studies have been undertaken at more than one scale within such hierarchies (Folt *et al.*, 1998). This raises an important issue where results from scientific research at one scale are transferred to management at another and has resulted in the development of Hypothesis (3).

Hypothesis (3): The scale of analysis (e.g. catchment, sub-catchment, reach) will influence the relationships identified between habitat controls and salmonid abundance and distribution. In other words, the scale of the control will be related to the scale at which its impact is observed.

It is also important to recognise that fluvial systems are dynamic and that processes such as channel migration, planform change, bank erosion, deposition, and bar formation are continually occurring. As such, fluvial systems are often considered to be in a state of dynamic equilibrium. The connection between these dynamic processes and ecology is also becoming increasingly being recognised. For example, floods, flow disturbances and bank erosion are essential to clean and replenish the gravels that are important as salmonid spawning and nursery habitat (Gilvear *et al.*, 2002). The importance of channel migration, bank erosion and planform changes has also been recognised as critical in controlling vegetation patch dynamics within riparian zones (Tiegs and Phol, 2005; Gilvear *et al*, 2002; Parsons *et al.*, 2006), and greater woody debris abundance has been observed downstream of active migrating channels (Piegay *et al.*, 2000). A final note regarding the importance of geomorphological and hydrological connectivity and coupling in determining in-stream habitat relates to the issue of landscape sensitivity as first raised by Brunsden and Thornes (1979). Well coupled systems transmit the effects of environmental change more readily than poorly coupled or buffered systems, and hence are more sensitive to change (Harvey, 2002). Therefore habitats that are well connected to the riparian zone and wider catchment will be more sensitive to pressures that occur within those zones.

2.7 Anthropogenic pressures on in-stream salmonid habitat

It is not just natural variables which influence the distribution and suitability of salmonid habitat. Anthropogenic actions can also have a profound effect upon hydrological and geomorphological processes. Just as natural controls can be viewed within a hierarchical framework, so can anthropogenic 'Pressures', and again it is hypothesised that the scale at which these pressures occur will influence the scale of impact upon salmonids. In the following sub-sections controls over in-stream salmonid habitat will be considered as occurring at the in-stream, riparian or catchment-scale.

2.7.1 In-stream pressures

In-stream pressures and modifications can have an immediate and severe impact upon salmonids, due to their immediate connection to in-stream habitat. Pressures can occur as a change to channel morphology, a change in flow regime or as an obstruction disrupting longitudinal connectivity.

2.7.1.1 Changes to morphology

Engineered pressures such as canalisation, hard-bank protection, dredging and embankments have been frequently implemented on many rivers within the UK for flood defence and navigation purposes, most commonly within urban areas, but smaller schemes have also occurred to protect agricultural land. The impacts of such modifications upon salmonid habitat are three-fold. First, they may lead to a reduction in channel migration and the renewal of channel bedforms. Riffles have been noted to exist either in a stable, fixed from with a compacted bed surface as relics of past processes, or as active features with 'fresh' surface sediments, little compaction and a pronounced bar front reflecting current sediment transport processes (Sear *et al.*, 2003) in actively meandering reaches. It is likely that active riffles represent prime spawning habitat. Second, modifications may lead to reductions in bankside cover which may have particular impacts upon trout. Third, they may lead to flow homogenisation and a loss of in-stream diversity. For example, riffle-pool sequences may be replaced by long reaches of run and glide, reducing habitat availability.

In 'Response' to such impacts a wide range of strategies have been implemented. These include the use of in-stream structures such as woody debris, flow deflectors, weirs and rubble mats to induce turbulence, direct flow and promote habitat diversity (Summers *et al.*, 1996; Hendry and Cragg-Hine, 1997). Such strategies have been used to great success particularly in Ireland, (e.g. the Lough Ennell enhancement programme O'Grady *et al.* (2002)). However, the authors do note that such schemes should be regarded as enhancement programmes, not rehabilitation schemes, as on-going maintenance is required. There are others who argue that a truly sustainable approach to river restoration should involve the removal of engineering structures, leaving the river to regain its natural state over time. However, such strategies are often not feasible where protection of property and infrastructure from flooding are required. In these cases active management may offer the only alternative.

2.7.1.2 Changes to flow regime

The flow regime of many rivers has been modified by in-stream pressures such as regulation and abstraction for water resources and hydro-electric power generation. The impacts of such schemes include, elevated and depressed low flows, diurnal fluctuations in flow, reduced flood frequency and magnitude (including total elimination) and unnaturally rapid rates of stage change (Gilvear *et al.*, 2002). This may have a number of impacts on salmonids (after Hendry *et al.*, 2003). First, it may modify water depth and wetted area impacting directly upon the availability of habitat. Reductions in depth may reduce the availability of holding pools which provide cover for adult fish and for juveniles during periods of high temperature. Reductions in velocity may reduce the degree of turbulence, lowering oxygen levels and reducing cover in riffles. Second, modified flows may directly affect channel morphology and therefore habitat availability. For example, a comprehensive study of morphological response to river engineering in Italy reported channel narrowing of up to 50%, a reduction in the braiding index, and channel incision (Surian and Rinaldi, 2003). The importance of flow variability is increasing being recognised as critical to the maintenance of channel morphology and ecological quality and is a function frequently lost in regulated rivers (Gilvear *et al.*, 2002). Third, modified flows may lead to modified sediment dynamics resulting in deposition of fines where flows are reduced and scour of spawning gravels where excessive flows are released. Fourth, a modified flow regime may also impact the availability of flows for salmonid migration limiting access to spawning grounds. Finally, flow modifications may also impact upon invertebrate abundance reducing food availability for salmonids (Gilvear *et al.*, 2002).

In recognising these impacts, many flow regulation schemes operate compensation flows. However, there is currently considerable debate as to the benefit of such releases in terms of the timing and level of releases that are critical to maintain ecological diversity (e.g. Gilvear *et al.*, 2002; Hendry *et al.*, 2003).

2.7.1.3 Obstructions to longitudinal connectivity

Just as obstructions such as weirs and dams can prevent the upstream migration of salmonids, they can equally prevent the downstream transport of sediment, causing water to become sediment starved. This can have severe consequences downstream of the obstruction, as water released from the dam possesses the energy to transport sediment, but has little or no sediment load. As such it is often referred to as 'hungry water' because the excess energy is typically expended upon increased bed scour and bank erosion downstream. This can lead to channel incision, changes in planform, a coarsening (armouring) of bed material and loss of spawning gravels as smaller gravels are transported without replenishment from upstream (Kondolf, 1997). Appreciation of the geomorphological significance of trapping the upstream sediment is now resulting in the use of substrate replenishment to prevent bed degradation and changes in substrate character downstream where removal of obstructions is not possible (Gilvear, 1999).

2.7.2 Riparian management

Riparian zones can be considered ecological boundaries, which physically separate terrestrial and aquatic ecosystems. As such they are important regulators of the movement of material through the catchment system (Burt *et al.*, 2002). Directly connected to the channel in many circumstances they are areas of high sensitivity, and pressures occurring in these zone may be transmitted readily and rapidly to in-stream habitats.

2.7.2.1 Riparian stock access and grazing

Within agricultural catchments such as the Eden, riparian functioning is often seriously impaired as a result of agricultural intensification, which has often concentrated on these zones due to their fertile floodplain soils. Of particular concern, stock access and grazing pressure within the riparian zone are often cited as major causes of salmonid habitat degradation. Indeed one scientific review into the impact of grazing stated that "No positive environmental impacts were found" (Belsky *et al.*, 1999). A number of negative impacts upon salmonid habitat have been described in the literature. First, the trampling of banks, particularly by cattle, has been shown to directly alter channel morphology. The shear-force applied by their hooves as cattle enter and exit the channel causes sediment to be dislodged and leads to the development of 'cow ramps' (trough shaped routes to and from the channel). During periods of high flow these areas are particularly vulnerable to further erosion as a result of water ingress into areas of increased hydraulic roughness and water turbulence. Where grazing is located on steep banks erosion can be particularly severe resulting in whole sections of bank collapsing into the channel (Trimble and Mendel, 1995). Bank slumping is a characteristic indicator of cattle trampling and it reduces the availability of undercut banks providing cover to salmonids (Belsky *et al.*, 1999). Removal of vegetation through grazing has also been shown to induce bank instability, and therefore increase susceptibility to erosion in areas of stock access. Accelerated erosion rates can result in channel widening, increased width:depth ratios, a reduction in water turbulence and the loss of pool-riffle sequences, which have been noted as an important component of salmonid habitat (Gilvear *et al.*, 2002). The effect of such changes may be especially significant during periods of low flow, when reduced depths can, in summer result in high water temperatures causing stress to salmonids; and in winter result in exposure of gravel bars and the consequential desiccation of redds and salmonid eggs (Hendry and Cragg-Hine, 1997). Second, accelerated erosion generates inputs of fine sediment which, depending on the extent of erosion may by locally or regionally significant (Walling, 2005), and could lead to the degradation of sensitive spawning habitats (Blesky *et al.*, 1999). Third, reductions in vegetation cover in turn reduce both cover for salmonids (Hendry *et al.*, 2003) and the input of terrestrial invertebrates which were discussed as an important food source especially for trout in upland rivers where autochthonous production is low (Gilvear *et al.*, 2002). Fourth, the capacity of riparian vegetation to act as a buffer towards suspended solids and nutrients can be reduced when it becomes depleted through grazing (Gilvear *et al.*, 2002). As reported by Meador and Goldstein (2003), decreased riparian condition across large geographic areas was associated with water quality constituents indicative of non-point pollution sources.

The widespread and now commonplace response to riparian degradation has been to exclude stock from the riparian zone through the use of stock-proof fencing, allowing vegetation to regenerate (Hendry *et al.*, 2003). Many projects have been undertaken showing impressive results such as, reduced width:depth ratios, the redevelopment of riffle-pool sequences, increased cover and invertebrate inputs. Localised improvements to salmonid populations have also been observed. For example, four years after 2km of stock exclusion fencing was established along the banks of the Todrig Burn, a tributary of the Ale Water within the Tweed catchment, a 1,232% and 292% increase in salmon and trout parr density respectively was recorded (Glen, 2002).

2.7.2.2 Riparian tree management

Tree management within the riparian zone is also considered by fisheries managers to be a significant issue (O'Grady, 2002). In Northern Europe during the 20th century riparian zones were frequently cleared of woodland. In such areas the original mixed assemblage of trees and shrubs has often been replaced by a monoculture of Alders (*Alnus glutinosa)* resulting in heavy shading and a 'tunnelling' effect. This has been associated with a reduction in in-stream productivity and hence food for salmonids, with effects becoming more significant as the length of tunnelled channel increases (O'Grady, 1993). In such areas coppicing has been promoted as a restoration strategy with removal of 50% canopy. However, as discussed previously, some overhead cover from riparian trees can also be beneficial. It therefore appears that there is currently little guidance as to the exact level of cover that is beneficial for salmonids of different species in different catchments (Armstrong *et al.*, 2003).

2.7.3 Catchment land use and management

The impact of catchment land use and land management on aquatic ecosystems is increasingly being raised as an important issue (e.g. Johnson and Gage, 1997; Allan and Johnson, 1997; Hendry *et al.*, 2003). Within the Eden catchment the predominant land use is agriculture (Mackay Consultants, 2003). Therefore, it is pressures associated with agricultural intensification that are discussed in more detail below.

2.7.3.1 Intensive grazing

Across much of the UK, stocking densities of both cattle and sheep have risen since the 1970s, largely associated with EU agricultural policies that have subsidised farmers on a per capita basis (Lane, 2003). Sansom (1999) reports that the total number of sheep increased by 40% between 1980 and 1993 in the UK, an increase that has also been accompanied by changes in stock management. Much of the UK uplands are now stocked all year round, reducing opportunities for re-vegetation. A decline in shepherding has also meant that sheep tend to congregate in localised areas causing grazing to become disproportionate. Intensive grazing by both sheep and cattle has been associated with changes to vegetation and the physical properties of soil such as, reduced biomass both above and below ground, soil compaction, increased bulk density and reduced pore space, resulting in changes to catchment hydrology such as reduced evapotranspiration, reduced infiltration rates, increased surface flow generation and higher peak flows (e.g. Langlands and Bennett, 1973; Gifford and Hawkins, 1978; Owens *et al.*, 1997; APEM, 1998; Greenwood *et al.*, 1998). Similar changes in peak flow have been observed downstream of urbanised areas and forestry where they have been associated with alterations to in-stream morphology such as channel deepening and widening (Gilvear *et al.*, 2002). Such changes may also occur downstream of intensively grazed areas resulting in a loss of salmonid habitat.

Intensive grazing has also been associated with increased soil erosion and fine sediment delivery as a result of soil exposure through loss of vegetation and increased surface runoff rates with greater stream power (Hendry *et al.*, 2003). For example, Owens *et al.* (1997) reported a reduction in soil loss from 2,259 to 9kg ha^{-1} following removal of cattle from grasslands. In the Lake District, sediment cores taken from Blelham Tarn showed an exponential increase in sedimentation rates correlated with increases in sheep densities (Van der Post *et al.*, 1997). The impacts of high stocking densities have been noted as particularly severe during winter when vegetation is dormant and opportunities for recovery are reduced. For example, Owens *et al.* (1997) reported that 60% of total soil loss from cattle grazed land occurred during the winter. Sediment delivery from stocked land may also be associated with the delivery of organic waste to the channel (Greig *et al.*, 2005). As discussed in section 2.5.1 increased fine sediment delivery and organic waste can have severe consequences for salmonid embryo survival within the redd, as well as reduced foraging ability and damage to gills for older life-stages.

2.7.3.2 Land drainage

Agricultural production is only possible in many areas as a result of land drainage using surface ditches or underground pipes to lower the water table and allow extended use of land previously waterlogged for much of the year. Drainage schemes were actively implemented across much of the UK during the 20th Century and have taken place both in lowland areas to allow cultivation of fertile riparian floodplains (Burt *et al.*, 2002) and in the uplands to improve grazing quality and for grouse shooting (Lane, 2003). The impact of land drainage is to modify the catchment's hydrological regime. By altering water table dynamics, changes to the spatial pattern of soil saturation may occur, altering water storage potential, infiltration dynamics, hydrological routing, and the timing of runoff responses to rainfall, ultimately modifying the downstream hydrograph with impacts for both flood flows and the maintenance of low flows. However, as noted in a number of papers, studies into the exact effects of land drainage have reported contradictory results (e.g. Lane, 2003; Hendry *et al.*, 2003). For example, Robinson (1986, *cited in* Lane, 2003) observed that the 90% flow exceedance doubled post drainage in an area of upland peat, whilst Newson and Robinson (1983, *cited in* Lane, 2003) found that drainage reduced peak flows and lengthened hydrograph duration in an area of peaty gley and podzol soils in Wales.

Land drainage may also impact water quality through effects on hydrological routing. Drains may increase hillslope to channel coupling bypassing the buffer zone functions of the riparian zone. Drained soils may encourage infiltration and reduce overland flow, leading to reduced soil erosion, the deposition of sediment and its associated contaminants. Whilst unsaturated, aerated conditions within floodplain soils may prevent anaerobic processes such as denitrification from operating (Burt *et al.*, 2002). Again the exact effects are noted to be complex and often contradictory.

2.7.3.3 Cropping regimes

In recent years a switch in the timing of arable production has been observed across the UK including in the Eden catchment, from spring to autumn sown cereals (Greig *et al.*, 2005), together with an expansion in maize production. This has in part been encouraged by subsidies such as the Arable Area Payments Scheme (Hendry *et al.*, 2003), but also by changes in climate, and modern farming techniques such as the use of chemicals and crop forcing under plastic which extend the growing season. These changes have been associated with an increase in fine sediment delivery as soils are now exposed and most vulnerable to erosion during autumn and winter coinciding with the typical period of highest rainfall in the UK. This switching in the timing of increased in sediment delivery also coincides with the period of salmonid spawning and egg incubation exacerbating the problem of fine infiltration and redd siltation (Greig *et al.*, 2005).

2.7.3.4 Chemical applications

Agricultural intensification has only been made possible in many circumstances through the development and application of chemicals. The use of fertilisers, herbicides, pesticides and insecticides is now widespread in the UK resulting in both point and diffuse pollution pressures. Of particular concern to fisheries managers, has been the use of synthetic pyrethroid (SP) sheep dip which, if inadequately disposed of, is significantly more toxic to aquatic organisms than the organophosphates (OP) products previously used (Hendry *et al.*, 2003). Within the Eden catchment there have been several incidents of SP pollution reported in one sub-catchment alone over recent years. In terms of diffuse pollution, many of the above substances can enter watercourses either through the process of leaching, or by becoming attached to soil particles which are subsequently eroded and transported via surface runoff. The impacts of enhanced nutrient concentrations and eutrophication as a result of fertiliser applications are widely documented (e.g. Heaney *et al.*, 2001). Laboratory experiments have shown many of the chemicals widely applied in agriculture to be toxic to fish, resulting in direct mortality at extreme levels but recent studies have also suggested many sub-lethal effects on fish physiology that may have severe consequences in the long-term. For example, low levels of cypermethrin (an SP insecticide) (>0.001 $\mu g\ l^{-1}$) have been demonstrated to affect the olfactory system in salmon, reducing the ability of male fish to detect and respond to the female priming pheromone at spawning resulting in a reduction in sperm production (Moore and Waring, 2001). Low levels of atrazine (herbicide) (0.04-14.0 $\mu g\ l^{-1}$) have shown similar effects, plus exposure of smolts to atrazine whilst in freshwater has been demonstrated to restrict their osmoregulatory capabilities resulting in 14-28% mortality on entering saltwater (Waring and Moore, 2004).

Whilst the above changes to land use and land management have been hypothesised to impact upon catchment hydrology and hence upon in-stream physical and chemical conditions, there is, to date, little scientific agreement upon the exact effects of these changes or that catchment land use is important in structuring ecological communities. Models for predicting fisheries carrying capacity such as HABSCORE (Milner *et al.*, 1993) tend only to include coarse variables such as the percentage of land under particular uses (e.g. agricultural and urban) within their parameter suite. As discussed in Chapter One, Section 1.4.2, the spatial distribution of hydrological processes such as infiltration, soil saturation, flow pathway and ultimately hydrological connectivity, as determined by the complex interaction of catchment variables such as geology, topography and soil type, may be just as important, if not more important, than the spatial distribution of land cover type in determining sensitivity of in-stream habitat to land management (Burt, 2001). It is these hydrological processes that ultimately determine the availability, mobilisation and transfer of water and contaminants from the catchment to the channel network. However, studies of these hydrological processes and the impact of land management upon them have rarely been undertaken at a catchment scale (Meador and Goldstein, 2003). For example, Lane (2003) notes that studies of upland gripping (drainage) have too often focused upon empirical studies of individual grips or small isolated networks, whereas at the catchment-scale it is the complex interaction of effects that ultimately determines downstream response. Similarly, studies of the effects of intensive grazing on fine sediment delivery or of nutrient leaching and buffer zone operation have often only been undertaken at the field scale. There is therefore a need to combine knowledge of the spatial distribution of land use and land management practices with knowledge of the spatial distribution of hydrological processes at the catchment-scale. As discussed in Chapter One, section 1.4.2, a progressive engagement between remotely sensed data and mathematical models is enabling science to make these links and to make statements about which locations in the landscape are likely to be causing habitat degradation (Lane *et al.*, 2006). Researchers have recently raised the possibility of extending the theory of hydrological connectivity to the catchment-scale as a tool for determining the impact of certain land management strategies upon the delivery of water, sediment and pollutants to the channel (e.g. Burt, 2001 and Lane *et al.*, 2003a), and a new environmental risk model, the *SCIMAP* model (Lane *et al.*, 2006) aimed at delivering this approach is currently under development and is to be used within this thesis.

Restoration strategies proposed for dealing with the impacts of catchment land management in general and agricultural intensification in particular, include riparian buffer zones and floodplain wetlands, drain blocking, nutrient budgeting, soil conservation measures and a change to more extensive management regimes in sensitive areas. However, without understanding as to whether land management influences fisheries at the catchment-scale and if so where, such strategies are likely to achieve at best minimal benefits. In the meantime, fisheries managers have focused on treating the symptoms of problems, for example, by gravel cleaning but such strategies are not likely to be sustainable or effective in the long-term.

2.8 DPSIR Synthesis

The aim of this chapter was to review and synthesise current understanding of the in-stream, riparian and catchment scale controls upon salmonid habitat. This has been achieved within a DPSIR framework (Figure 2.5). To begin with, a review of current ecological understanding regarding salmonid population dynamics was undertaken to examine the mechanisms by which salmonid populations are regulated or limited. This highlighted the importance of the environment in determining 'carrying capacity' and influencing population dynamics whether actively by stimulating competition and predation or passively, for example, through starvation or asphyxiation. Habitat availability and heterogeneity was shown to control carrying capacity and also the environment's resilience or sensitivity to both anthropogenic and natural pressures. However, it was also noted that salmonid mobility according to life-stage may be important in determining the scale of habitat heterogeneity required or alternatively, the scale over which pressures must apply to have an impact. It was then critical to identify which components of the environment 'State' are ecologically relevant to salmonids so that these in turn can be used to identify which hydrological and geomorphological processes are most likely to result in ecologically relevant impacts. It has been recognised that different components are relevant at different stages in the life-cycle and for different species. As such, the 'State' section of the DPSIR framework has been separated into spawning, fry and parr habitat requirements. Following this it was hypothesised that different pressures may be important in different locations specific to the species, life-stage and scale of habitat in question. This confirmed the approach outlined in Chapter One which stated the need for a catchment-wide analysis of salmonids and their habitat that is spatially explicit. The review then focused on current hydrological and geomorphological understanding to identify both natural and anthropogenic 'Drivers and Pressures' (controls) which influence the identified, ecologically relevant, components of the freshwater environment. Here, the role of scale was recognised as critical, both in terms of the scale of the pressure in determining the scale of its impact, but also the scale of investigation in identifying which scale of pressure has the greatest control over salmonid abundance and distribution. This emphasised the need to evaluate the relationships between habitat and salmonid populations within a hierarchical framework as presented in Chapter One (Figure 1.4).

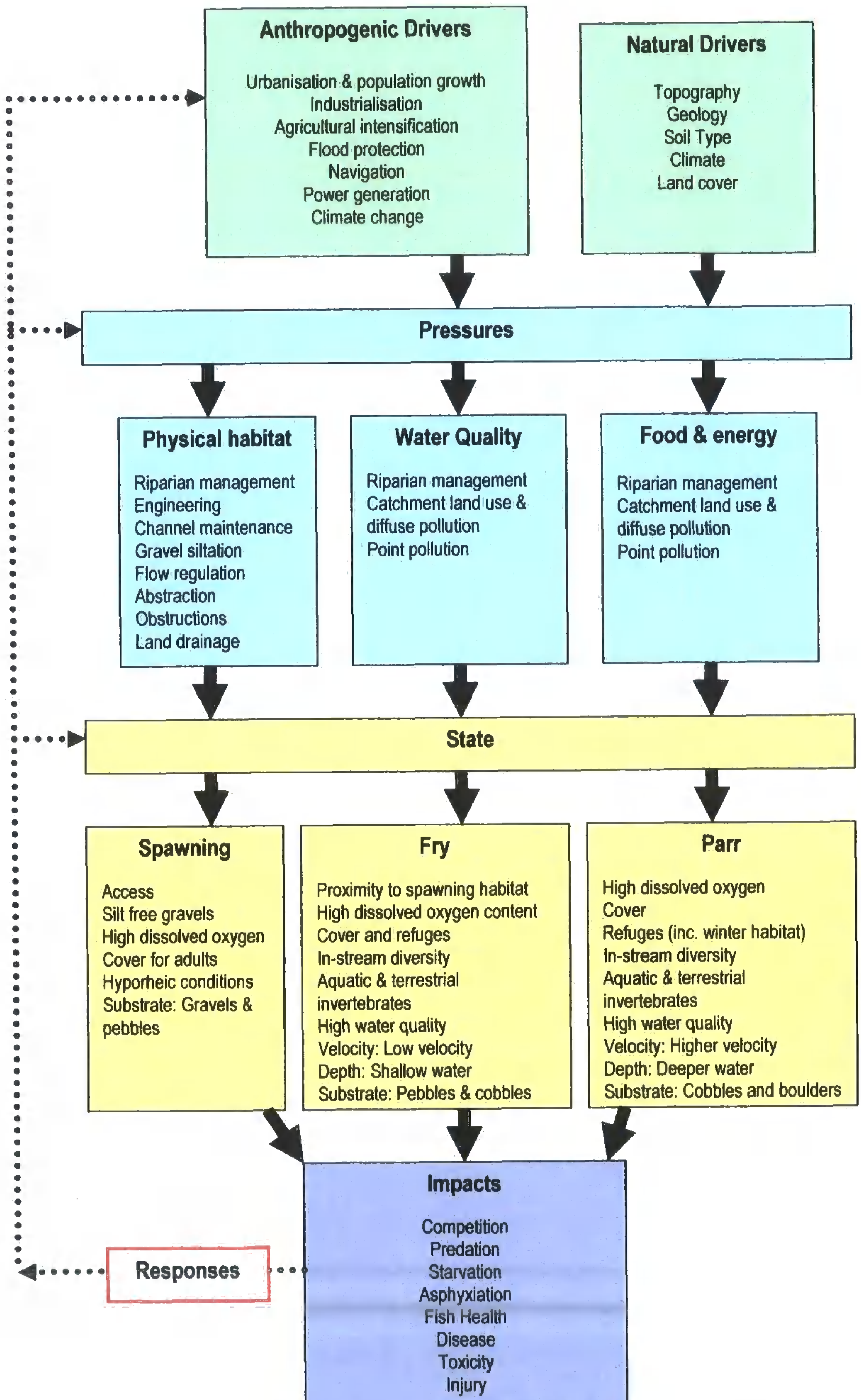


Figure 2.5: *Conceptual DPSIR framework of the controls on freshwater salmonid habitat and salmonid abundance.*

During the review a number of hypotheses have been identified for further investigation within the thesis.

Hypothesis (1): Relationships between habitat and salmonid abundance and distribution are structured by life-stage according to the level of mobility and potential for dispersal at each life-stage.

Hypothesis (2): Relationships between habitat and salmonid abundance are species and location specific relating to the scale of habitat occupied by different species.

Hypothesis (3): The scale of analysis (e.g. catchment, sub-catchment, reach) will influence the relationships identified between habitat controls and salmonid abundance and distribution. In other words, the scale of the control will be related to the scale at which its impact is observed.

These will be assessed in Chapter Six of the thesis by relating spatially distributed data on salmonid abundance to spatially distributed data on habitat controls and habitat indicators. This will be a catchment-wide assessment undertaken in a hierarchical framework. Not all the pressures identified in Figure 2.5 are to be assessed. Instead only those considered most relevant to the Eden catchment, a predominantly rural agricultural landscape have been chosen. Pressures have been selected from all three scales (in-stream, riparian and catchment) and include both natural controls (e.g. channel slope, flow type, substrate, and catchment-channel connectivity risk), and anthropogenic pressures (e.g. obstructions to migration, gravel siltation, stock access and catchment land use). A detailed list of habitat variables is presented in Chapter Three (Table 3.1)

Chapter Three - Data sources for the Eden catchment

3.1 Introduction

Chapter Two synthesised current understanding of in-stream, riparian and catchment-scale controls on salmonid habitat using a DPSIR framework (Figure 2.5), and three hypotheses regarding the relationship between these controls and salmonid populations were proposed. Successful investigation of these hypotheses is dependent on the acquisition of high-resolution spatial data, both on habitat controls and salmonid populations at the relevant scales. Acquiring such data has been a major obstacle to this type of research in the past. Traditional data collection methods typically consist of site-based sampling at small spatial scales (e.g. reach scale), focused on in-stream and riparian controls (Folt *et al.*, 1998; Thompson and Lee, 2000; Pess *et al.*, 2002). These techniques have been employed to develop and validate a number of well established site-based models for relating salmonid populations to habitat conditions. These include: (1) HABSCORE (Milner *et al.*, 1993) which predicts salmonid abundance from habitat features such as mean depth, mean width, substrate embeddedness, substrate diversity, flow type and discharge range at scales of 10-250m; and (2) PHABSIM which combines hydraulic simulations of habitat (depth and velocity) with indices of habitat suitability (for depth, velocity and substrate) to quantify the availability of habitat as a function of discharge (Gibbins *et al.*, 2002). Scaling up to analyse relationships between salmonid populations and habitat at the catchment-scale incorporating catchment-scale controls has until now proved impractical and prohibitively costly. However, technological advancements in computing power, global positioning systems (GPS), Geographical Information Systems (GIS), remote sensing and image processing now enable quantitative assessment of many environmental components at a catchment scale (Johnson and Gage, 1997).

Central to allowing larger scales of analysis is the growing range of regional and national datasets now widely available, providing information on catchment-scale variables such as land cover, climate, soil type, geology and topography, many downloadable from the internet. This is in part being driven by pressure on governmental bodies and agencies to make data more accessible to the public. For example, in the UK, a number of governmental agencies have collaborated to develop the MAGIC web-site (www.magic.gov.uk) including data on administrative boundaries, environmental schemes and designations, which are freely downloadable to the public. Within the academic arena, an even greater number of datasets including digital terrain models (DTMs), satellite imagery, digital map data, and climatic information are freely available for *bona fida* research through a range of data providers such as Edina (http://edina.ac.uk); the Natural Environment Research Council (NERC) Earth Observation Data Centre (www.neodc.rl.ac.uk/) and UK Climate Impacts Programme (UKCIP) (www.ukcip.org.uk). Whilst valuable data sources in their own right, the true power of these datasets becomes most apparent when they are combined with environmental models, or subjected to advanced GIS and image processing techniques, to create powerful tools that can: (1) derive secondary variables of interest such as channel slope, and soil erosion risk; (2) quantify spatial patterns, position and spatial relationships (e.g. shape, diversity, proximity, connectivity and juxtaposition); and (3) analyse and predict the impacts of catchment-scale processes on environmental and ecological phenomenon. GIS technology further enables integration of catchment-scale data with that collected at smaller spatial scales to test hierarchical theories and investigate interactions between scales assessing their relative importance at different locations within catchments (Johnson and Gage, 1997).

As noted above, this thesis is heavily reliant on the ability to acquire high-resolution spatial data on habitat controls at a range of scales. Many of these data are derived from widely available digital GIS datasets and/or remotely sensed data specifically collected for the project purposes. In addition, high-resolution spatial data on salmonid abundance are also required at the catchment-scale. The aim of this chapter is to identify the primary datasets required and to review the catchment-wide data sources that are available for the Eden to fulfil these needs. The methods by which data have been collected will be discussed, together with an evaluation of their accuracy and the uncertainty involved in their use. The data identified and discussed within this chapter will then be used in Chapters Four and Five to derive specific habitat variables to achieve the requirements of Objective 2 of this thesis: *To employ recent advances in remote sensing, GIS and environmental modelling, to identify, to develop and to validate tools for quantifying salmon and trout habitat at the catchment-scale, appropriate to each habitat control and scale of control.*

3.2 Features of Interest

The first stage of any ecological assessment is to identify the features, or in this case habitat controls, of interest that are to be investigated. Following Chapter Two, a number of the key controls identified within the DPSIR framework, both natural and anthropogenic, have been selected from each scale as potentially relevant to the Eden catchment (Table 3.1), which is predominantly rural and agricultural. The next stage is to determine how information regarding these features may be acquired. To this end, Table 3.1 highlights potential methodologies for acquiring data based on the technological developments mentioned above together with their primary data requirements.

In summary, a number of primary data requirements have been identified as follows:

- Catchment-wide survey of salmonid abundance
- Digital aerial photography of the channel and riparian zone
- Digital terrain model
- Land cover data
- Rainfall data
- Impassable barrier locations

Before potential data sources for each of the specific requirements can be evaluated, there are a number of general conditions that should be considered, which will apply to all data as follows:

1. **Catchment-wide coverage:** As previously acknowledged, successful fisheries management can only be achieved with a catchment-wide approach, particularly when considering highly mobile species such as Atlantic salmon and brown trout which make extensive use of the river system throughout their life-cycle. Achieving management at this scale relies on data that are also spatially distributed to generate catchment-wide coverage, as different controls may be operating in different locations to different extents.

2. **Spatial resolution:** Careful consideration should be given to the spatial resolution of data which, within the confines of available data, should be fine enough to capture spatial heterogeneity in habitat controls. Where the necessary data are not available sub-resolution treatments may be required.

3. **Spatially referenced:** The ability to integrate data from different sources and facilitate multivariate analysis is essential to the aims of this research. This relies on data being accurately and precisely geo-referenced to ensure that data from one source can be readily associated with data from another source at the correct location.

4. **Digital data:** In conjunction with (3), data are to be integrated and analysed using GIS. Therefore, the data must already exist in, or be readily convertible to, a digital format compatible with the software package used. The primary GIS system used in this research is ArcGIS v.9.

Table 3.1: *A summary of habitat controls and indicators together with data collection methodologies and data requirements.*

Variable		Details	References	Methodology	Data Required
In-stream controls	Impassable barriers	Barriers to migration both artificial (e.g. weirs, dams and culverts) and natural (e.g. waterfalls) can lead to habitat fragmentation and a loss of longitudinal connectivity.	Calles and Greenberg (2005)	Walkover survey to map and geo-reference all known barriers.	▪ Geo-referenced locations of all known barriers
	Channel gradient	Salmonid distribution may be related to channel slope which influences in-stream hydraulic conditions and the availability of in-stream habitat types.	Hicks and Hall (2003)	Digital terrain model (DTM) processing within GIS to extract channel gradient	▪ DTM
	Hydraulic conditions	Throughout their life cycle salmonids utilise areas of different hydraulic flow conditions tending to move towards deeper slower habitat as they grow.	Armstrong *et al.* (2003)	Automated classification of relative water depth form high resolution aerial photography. Reflectance varies with depth.	▪ High resolution aerial survey of the channel and river corridor.
	Flow type	Physical biotopes (e.g. riffles/pools/glides) are functions of the hydraulic conditions at a given site which may be represented by channel gradient. They may be used to segregate the channel into identifiable units which have ecological meaning in terms of salmonid habitat.	Padmore (1998) Marcus *et al.* (2003) Sear *et al.*,(2003)	Classification of flow type either from aerial photography based on reclassification of the above depth classifications, or from re-classification of channel slope.	▪ High resolution aerial survey ▪ DTM
	Channel substrate	The size of substrate required varies throughout the life-cycle. Substrate condition is also important (e.g. siltation and concretion)	Armstrong *et al.* (2003)	Manual interpretation of high resolution aerial photography to map broad substrate types.	▪ High resolution aerial survey
Riparian controls	Riparian land use	Local riparian land use can impact on in-stream conditions by determining bankside vegetation cover, influencing bank structure and strength and regulating pollutant inputs.		Manual interpretation of high resolution aerial photography to map riparian land use type	▪ High resolution aerial photography
	Stock access	Agricultural stock access within the riparian zone can lead to overgrazing, a reduction in vegetation cover, increased catchment sediment sources, damage to bankside structure and strength and accelerated bank erosion.	Trimble and Mendel, (1995).	Manual interpretation of high resolution aerial photography to identify channel reaches accessed by stock.	▪ High resolution aerial photography
	Bank erosion	Accelerated erosion can lead to increased sediment delivery, bank collapse, increased width:depth ratios, over-widening and the degradation of pool-riffle habitat. However, natural levels of fluvial erosion may create habitat through the provision of undercut banks and exposing tree roots.	Belsky *et al.* (1999); Trimble and Mendel, 1995; Hendry *et al.* 2003.	Manual interpretation of aerial photography to locate areas of active erosion and identify the cause (e.g. stock, fluvial, topographic) and severity of that erosion.	▪ High resolution aerial survey
	Shade and cover	It has been suggested that complete shading of 400m can result in the absence of salmonids. The effects of bankside trees are contentious as benefits such as bank stabilisation and tree root cover are also cited and should also be considered.	O'Grady (1993) Summers, (2002)	Manual interpretation of high resolution aerial photography recording tree density and % channel coverage within the GIS.	▪ High resolution aerial survey

Table 3.1 cont...

Variable		Details	References	Methodology	Data Required
Catchment controls	Land cover	Water quality parameters such as gravel siltation, nutrient levels and toxin concentrations will impact on habitat quality. The type of catchment land cover will influence the degree of pollutant production and availability within the catchment.	Armstrong *et al.*, (2003) Wang *et al.*, (2003) Pess *et al.*, (2002)	Risk based classification of land cover in relation to the likely intensity of use and production of pollutants.	▪ Catchment land cover
	Catchment-channel connectivity	The risk that pollutants are delivered to the channel network is determined by the type and level of hydrological connectivity that exists between the landscape and channel network.	Lane *et al.*, (2006) Burt and Pinay (2005)	Application of the *SCIMAP* model to determine the probabilistic risk of catchment-channel surface hydrological connectivity.	▪ DTM ▪ Rainfall data
Target variable	Salmonid populations	Catchment-wide data on salmonid population abundance and distribution are required in order to investigate relationships with habitat variables. A wide coverage is required due to the non-uniform distribution of populations that may be misrepresented by only sampling at a few sites.	Crozier and Kennedy (1994)	Semi-quantitative electrofishing of fry populations with additional fully quantitative sampling for parr populations at a selected number of sites.	▪ Catchment-wide high resolution survey of salmonid abundance.

5. **Continuous data:** As far as possible, continuous habitat data should be collected. Sampled data may characterise tributaries, sub-catchments and the whole catchment adequately enabling relationships between habitat and salmonid populations to be studied, but such data will not facilitate precise location of management issues for targeting purposes.

6. **Up to date data:** Rivers and catchments are highly dynamic systems that are constantly changing. Successful habitat restoration therefore requires management decisions to be based upon current and reliable data sources about the controls that are currently operating at each location.

7. **Rapid and cost-effective collection:** Managers typically have to make decisions within time pressured deadlines and have limited funds for undertaking restoration projects. The more rapidly and cost effectively they can collect data, the more time and money they have available for actually carrying out projects. However, wherever possible, data quality in terms of the above requirements should not be sacrificed in order to meet this need.

Each of the specific data requirements is now dealt with in turn.

3.3 Salmonid population data

Catchment-wide information regarding the spatial variability in salmonid abundance and distribution is a fundamental requirement of this research. Salmonid populations typically exhibit a high degree of spatial variability reflecting the underlying spatial variability in habitat availability and quality (Williams and Hendry, 2003). The ability to capture this variability depends on the spatial resolution of data available. Various techniques exist for assessing fish populations, and the technique chosen must be applicable to the species, life-stage, habitat, spatial resolution and scale of interest. Table 3.2 reviews the main techniques available in terms of their suitability for assessing salmonid populations at the catchment-scale as required by this research.

Table 3.2: *A summary of fish monitoring techniques*

Technique	Target Population	Description	Suitability to this research	References
Fish Counters	Smolts Adults	Electronic sensors used to count or photograph fish as they pass. Particularly suited to use in channel narrowings (e.g. fish passes), they are used to monitor fish stocks entering or exiting rivers on migratory runs.	Expensive and limited to a small number of sites. Does not capture spatial variability.	Thorley *et al.* (2005)
Fish Trapping	Smolts Adults	Usually undertaken to monitor stocks during migratory runs, trapping is used to assess smolt or adult recruitment for individual catchments or sub-catchments.	Expensive and limited to a small number of sites. Traps generally require daily operation. Does not capture spatial variability	Crozier and Kennedy (1995)
Fish Tagging	Parr Smolts Adults	Tagging methods included PIT tags, and radio-tagging. Typically used to monitor fish movements from micro-habitat selection and diurnal movements to migratory paths and spawning destinations,	Expensive and limited to a small number of fish Only suitable for parr and older fish due to the size of tags	Riley *et al.* (2006)
Snorkel Surveys	Parr Adults	Expert surveyors swim upstream within pools and glides and count the number of fish observed within a sample reach (e.g. 100m stretch). Can be used to observe micro-habitat selection by fish, or provide estimates of population density.	Rapid and inexpensive allowing high resolution spatial data to be collected. Relies on the expertise of the individual diver. Not suitable for surveying in riffle (fry) habitats. Fish may be missed or counted more than once. Visibility can be impaired by turbidity, low light intensity, water turbidity, aquatic macrophytes and woody debris.	Thompson and Lee (2000) Joyce and Hubert, (2003)
Catch Returns	Adults	Anglers record the number of fish caught, length, weight and most importantly time spent fishing. This enables catch per unit effort to be calculated.	Data are often sparse and of limited quality. Dependent on the accuracy with which individual anglers record data.	
Redd Counts	Spawning	Expert field surveyors undertake daily monitoring of selected sites during the spawning season. Typically used to assess spawning site selection and estimate spawning density.	Reliant on surveyor expertise and spatially limited to a few kilometres. Does not provide information on spawning success.	Webb *et al.* (2001); Gibbins *et al.* (2002)
Electrofishing (Quantitative)	Dependent on the habitat targeted, typically juveniles	Reaches, typically 50m in length, including both riffle and pool habitats are stop-netted and electrofished. Multiple pass depletion or mark and re-capture methods can be used. Typically used to provide estimates of fish density at a site.	Time consuming and therefore restricted to precise point estimates at a few sites. Does not capture spatial variability	Crozier and Kenedy (1994)
Electrofishing (Semi-quantitative)	Fry (0+)	Operatives fish for a standard period of time (typically 5 minutes) within riffle habitat. No stop-nets are required. The number of fish caught in 5 minutes can be converted into an estimate of density using calibration equations based on quantitative fishing at a sample survey sites. Typically used to provide data on the spatial variability in fish populations.	Rapid and inexpensive allowing high-resolution spatial data to be collected. Not suitable for surveying parr due to their greater mobility. Less precise than quantitative methods.	Crozier and Kennedy (1994)

Of the techniques included in Table 3.2, only semi-quantitative electrofishing and snorkel surveys enable spatial variability in populations to be captured cost-effectively at a high resolution. For example, 6-8 semi-quantitative electrofishing surveys may undertaken in a day, enabling between 250-350 sites to be surveyed in a summer field season, with a spatial resolution of approximately one site per kilometre of river (pers comm.J.Brown). All the other techniques are constrained, primarily by expense, to small spatial scales, and may be misleading as a catchment-scale indicator due to the non-uniform distribution of salmonids. Semi-quantitative electrofishing has been selected here as it is considered a more objective survey technique than snorkel surveys which are heavily reliant on the expertise of the individual surveyor. In addition, fish may be missed, counted more than once, or incorrectly classified as salmon or trout during snorkel surveys, particularly where visibility is impaired due to turbidity, light intensity, turbulent water, aquatic macrophytes or woody debris. Additionally, they are not suitable for surveying riffle habitat. As such snorkel surveys are most suited to large clear streams with little cover (Zubik and Fraley, 1988). These environmental limitations severely restrict the ability of snorkel surveys to assess salmonid abundance in relation to a variety of habitat conditions, many of which were noted in Chapter Two to be critical components controlling the quality of salmonid habitat. Semi-quantitative electrofishing is therefore the only technique currently available that can provide objective, high-resolution spatial data on salmonid populations at the catchment-scale. The technique was developed by Crozier and Kennedy (1994) and has been successfully applied by a number of studies (e.g. Tiffan *et al.*, 2002; Wyatt, 2002).

3.3.1 Semi-quantitative electrofishing

Semi-quantitative electrofishing involves two operatives fishing in a downstream direction for a fixed period of time, typically 5 minutes. Single anode backpack electrofishing gear is used and no stop nets are required (Figure 3.1). The technique is only suitable to surveying fry. Parr and adult salmonids may be caught but their higher mobility makes catching efficiency lower in the absence of stop nets and may not be representative of actual densities (Crozier and Kennedy, 1994). However, fry are considered an ideal life-stage to use here as this research is attempting to evaluate the impact that in-stream, riparian and catchment-scale controls have on habitat and therefore salmonids at discrete spatial locations within the channel network. As the least mobile life-stage, it is valid to make the assumption that fry populations are strongly influenced by local conditions. Research by Milner *et al.*, (1995) has also suggested the temporal variance displayed by 0+[4] salmon and trout is consistently lower than that observed in older populations. It is therefore likely that fry represent the best indicator of spatial variance which is the key focus of this thesis. As many 0+ salmonids as possible are caught in a dip net by the operative who is positioned downstream of the anode and held (with aerators if necessary) until the end of fishing. They are then separated into species and measured to the nearest mm (fork length) (Figure 3.2). Fish are then returned to the river. Length-frequency relationships are used to separate 0+ and older fish, and numbers of 0+ salmon and trout are expressed as number caught per 5 minutes fishing (n $5min^{-1}$), classified into 5 grades (Table 3.3), (Crozier and Kennedy, 1994). Site location is recorded using a handheld GPS (e.g. Garmin GPS12) to enable GIS integration. It is important to recognise that sacrificing time to gain better spatial resolution results in enhanced spatial variance or sampling enhanced noise (Wiley *et al.*, 1997). In an attempt to reduce some of this spatial noise, semi-quantitative electrofishing surveys are stratified to only those sites where fry are likely to be found (riffle habitat). Further, only those sites where fishing efficiency is estimated to be greater than 60% are retained for analysis. Efficiency is crudely estimated by counting the number of 0+ salmonids seen but not captured during the 5 minutes and dividing the number caught by the total number seen.

Figure 3.1: *Semi-quantitative electofishing in action! Photograph courtesy of A.Tryner, and the Countryside Agency*

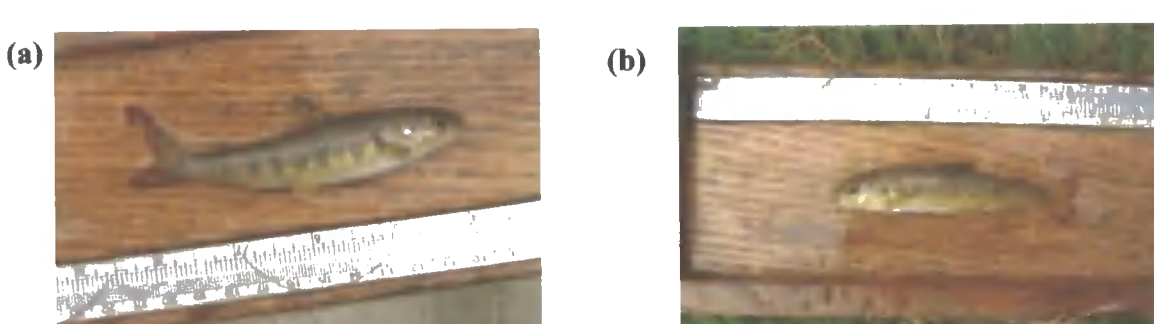


Figure 3.2: *Fish are identified as (a) salmon or (b) trout, counted and the fork length measured before returning them to the river. Photographs courtesy of J. Brown*

[4] 0+ refers to fish in their first year of life

Table 3.3: *Fry classification scheme used by Eden Rivers Trust, based on distribution percentiles for (0+) salmon obtained from quantitative electrofishing in Ireland (Crozier and Kennedy, 1994).*

Density Classification	Semi-quantitative ($n5min^{-1}$)
A (excellent)	>23
B (good)	11-23
C (fair)	5-10
D (poor)	1-4
E (absent)	0

In the Eden catchment semi-quantitative electrofishing has been carried out annually by the Eden Rivers Trust (ERT) since 2002, surveying between 250 and 350 sites per year (e.g. Maltby, 2002; Townsend-Cartwright, 2003; Dickson, 2004; Brown, 2006a&b) (Figure 3.3 and Appendix 1). The Environment Agency (EA) also carries out routine electrofishing surveys within the Eden catchment, comprising 25 quantitative annual monitoring sites, together with up to 50 rotational sites. This programme was designed to provide data for national reporting requirements on the status of salmonid stocks and, as identified by ERT, does not provide data of sufficient spatial resolution to target fisheries management at the catchment-scale (Williams and Hendry, 2003). Calibration between sites surveyed by ERT and the EA allows pooling of data to enhance spatial resolution (Townsend-Cartwright, 2003). All surveys are carried out to standards developed by the European Standards Committee (CEN, 2001), the Environment Agency and APEM Ltd by an experienced fisheries scientist assisted by the help of trained volunteers. Data collected in 2002 were subject to detailed statistical assessment to design and check the random stratified sampling protocol used in later years (Williams and Hendry, 2003). The aim was to determine the sample size necessary to characterise trout and salmon fry populations at a catchment-scale, with emphasis given to capturing the spatial variance within a given year. This was calculated using Bohlin's catch per unit effort methodology (Bohlin *et al.*, 1990) and a Class 3 level of determination (sufficient to detect a 50% change in populations) applied at the catchment, area (Figure 1.6a) and tributary level. Analysis indicated that 66 and 347 samples were required to characterise salmon and trout fry populations respectively at the catchment-scale. Trout fry exhibit much greater spatial clustering within the Eden catchment than salmon fry which are more evenly distributed. Samples were then apportioned in a stratified manner between areas and tributaries according to the level of spatial variance associated with each. Within each tributary, sites are sampled randomly with no repetition between years. However, additional trout sites required in excess of those used to monitor salmon fry should be located within areas only likely to be inhabited by trout (Williams and Hendry, 2003).

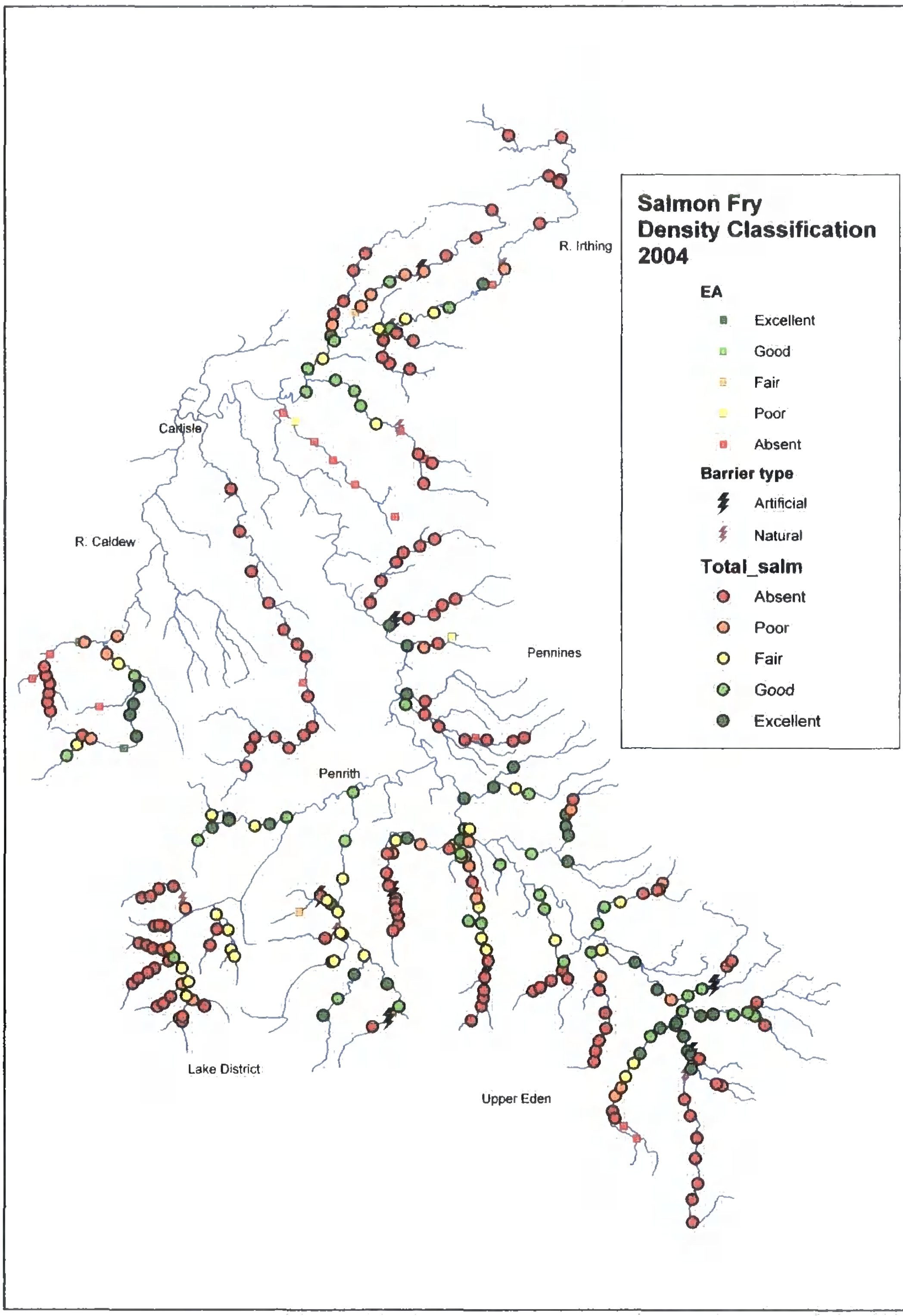


Figure 3.3: *Catchment-wide coverage achieved with semi-quantitative electrofishing. Salmon fry populations surveyed by the Eden Rivers Trust in 2004 (Reproduced from Dickson, 2004).*

3.3.2 Rapid quantitative cluster electrofishing

As discussed, semi-quantitative electrofishing is only applicable to salmonid fry. However, as identified in Chapter Two, investigating Hypothesis (1), "*Relationships between habitat and salmonid abundance and distribution are structured by life-stage according to the level of mobility and potential for dispersal at each life-stage,*" relies on the ability to obtain data at more than one life-stage. For this reason, a programme of quantitative electrofishing surveys was undertaken by ERT during 2005 aimed at capturing data on parr populations within an area of the upper catchment. Quantitative electrofishing surveys were undertaken in reaches (including both riffle and pool habitat) of approximately 30-50m in length that had been stop-netted. Sites were fished using a multiple-pass, triple shock technique. Operatives fished in an upstream direction using single anode, bankside generator-driven gear, and covered the entire stop-netted area (Figure 3.4). As the population is trapped, provided effort is constant and the population not too large, each run should result in a decrease in the number of fish caught. This allows depletion statistics (e.g. De Lury, 1947; Carle and Strubb, 1978) to be applied, the rate of depletion determining the population estimate for the reach (Williams and Hendry, 2003). A 30-minute break is allowed between each pass. Fish are retained and processed separately after each pass, and are only returned to the river upon completion of the site. As for fry, species and fork length were recorded, together with the approximate wetted area of the site, and GPS location. Population ($N100m^{-2}$) estimates at these sites were calculated by ERT using a Carle and Strubb (1978) regression. As highlighted in Table 3.2, quantitative electrofishing is restricted to precise point estimates at a few sites due to the increased time required for surveys. More cumbersome gear needs four as opposed to two operatives. To enable wider spatial coverage to be achieved a rapid quantitative cluster technique has been developed by APEM Ltd (*pers. comm.* K. Hendry). This enables fully quantitative sites to be related to a number of (typically 2-4) cluster sites. Cluster sites are stop-netted but only fished with a single pass. They must be representative in terms of habitat to the quantitative site, for example, all upland step-pool environments generally located within a few kilometres of each other. The depletion rate calculated from the fully quantitative site is then applied to the cluster sites to again estimate the total number of fish present in $100m^2$. Using this technique ERT was able to survey 54 sites for parr (16 fully quantitative and 38 cluster sites) (Brown and Dugdale 2006).

Figure 3.4: *Quantitative electrofishing in action, using stop-nets and bankside generator-driven gear. Photographs courtesy of J. Brown.*

3.3.3 Calibration of semi-quantitative electrofishing

Semi-quantitative surveys can be calibrated to provide estimates of fish abundance (n $100m^{-2}$) using quantitative electrofishing at a sample of sites (Crozier and Kennedy, 1994). For the purposes of calibration, a 5-minute semi-quantitative survey, only targeted at riffle habitat, is completed within the stop-netted section prior to depletion fishing. As in the cluster technique, results from semi-quantitative electrofishing are then related to quantitative results using regression analysis to convert n $5min^{-1}$ to n $100m^{-2}$ (Table 3.4, Figure 3.5). As the Environment Agency use a six-tiered classification compared to Crozier and Kennedy's five-tiered system calibration allows conversion of EA data so it can be combined with ERT results (Brown, 2006a).

Table 3.4: *Semi-quantitative electrofishing and its relationship to actual fish densities derived from quantitative electrofishing (Brown, 2006a).*

Density Classification	**Semi-quantitative**	**Quantitative equivalent (N$100m^{-2}$)**	
	($n5min^{-1}$) Salmon & Trout (Crozier & Kennedy)	Carle & Strubb (Eden) for 0+salmon	Carle & Strubb (Eden) for 0+trout
A (excellent)	>23	>60.9	>132.3
B (good)	11-23	29.1-60.8	63.3-132.2
C (fair)	5-10	13.2-29.0	28.7-63.2
D (poor)	1-4	2.6-13.1	5.8-28.6
E (absent)	0	0-2.5	0-5.7

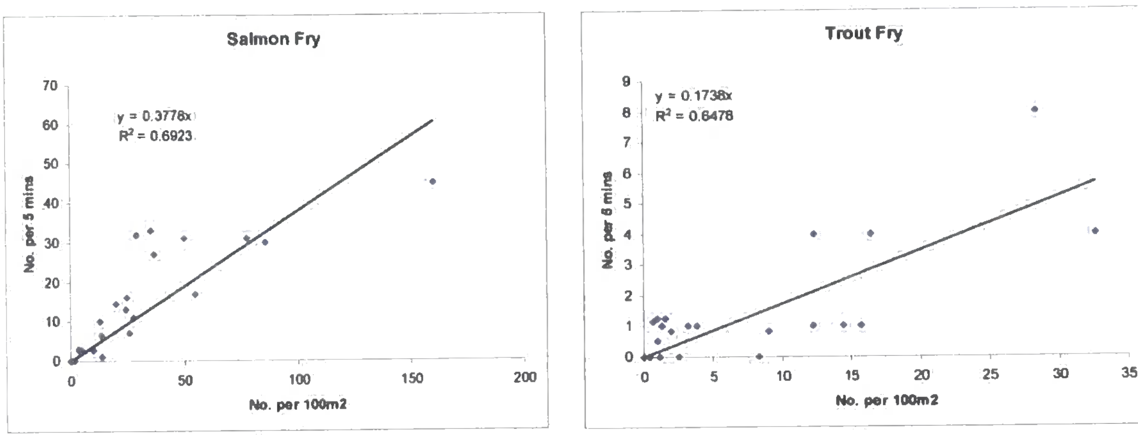


Figure 3.5: *Calibration of semi-quantitative electrofishing, Brown, (2006a).*

Table 3.4 suggests that within the Eden catchment semi-quantitative electrofishing is less efficient for trout fry than for salmon. This may be related to differences in the in-stream habitat occupied by the two species and/or their specific response to electricity. To standardise the classification system between the two species the 5-minute classes were re-coded for trout so that A to E equalled the same number of fry in 100m² as for salmon, based on ERT calibration data for 2005 (Table 3.5).

Table 3.5: *Standardisation of the 5-tier salmonid fry classification system between salmon and trout*

Density Classification	Quantitative fish (n100m^{-2})	Semi-quantitative	
	0+ salmon & trout (from ERT salmon fry calibration data)	(n5min^{-1}) Salmon (Crozier & Kennedy)	(n5min^{-1}) Trout (recoded based on ERT calibration data)
A (excellent)	>60.9	>23	>11
B (good)	29.1-60.8	11-23	5-11
C (fair)	13.2-29.0	5-10	2-4
D (poor)	2.6-13.1	1-4	1
E (absent)	0-2.5	0	0

3.3.4 Additional data

In addition to data on salmonid numbers, a range of other variables were recorded at each electrofishing site during the 2004 and 2005 field seasons, including estimates of other species observed, riparian condition, channel substrate, gravel siltation and bank erosion (Appendix 1). I assisted the ERT fisheries scientist for a four week period during 2005 and gained experience of both semi-quantitative and quantitative electrofishing methods. This experience enabled me to directly observe habitat utilisation by salmonids under a diverse range of habitat conditions, including microhabitat selection within survey reaches, and this has helped with interpretation of results throughout this thesis. Electrofishing data were provided by Eden Rivers Trust as a Microsoft Access database, which was converted to ArcGIS point shapefiles using the GPS coordinates recorded in the field.

In addition to electrofishing data, the Environment Agency also operates a resistivity fish counter at Corby on Eden (OS Ref: NY 469545). Resistivity counters sense the presence of a fish by detecting the change in electrical resistance of the surrounding water (Thornley *et al*., 2005). This counter provides data on the potential number of adult salmon broodstock returning to the River Eden catchment each year. Figure 3.6 highlights the temporal variation observed in returning salmon to the Eden catchment. Due to this temporal variation, data from the electrofishing surveys should not be combined and instead, analysis of spatial distributions and relationships between salmonid populations and habitat throughout this thesis will be conducted in terms of individual years.

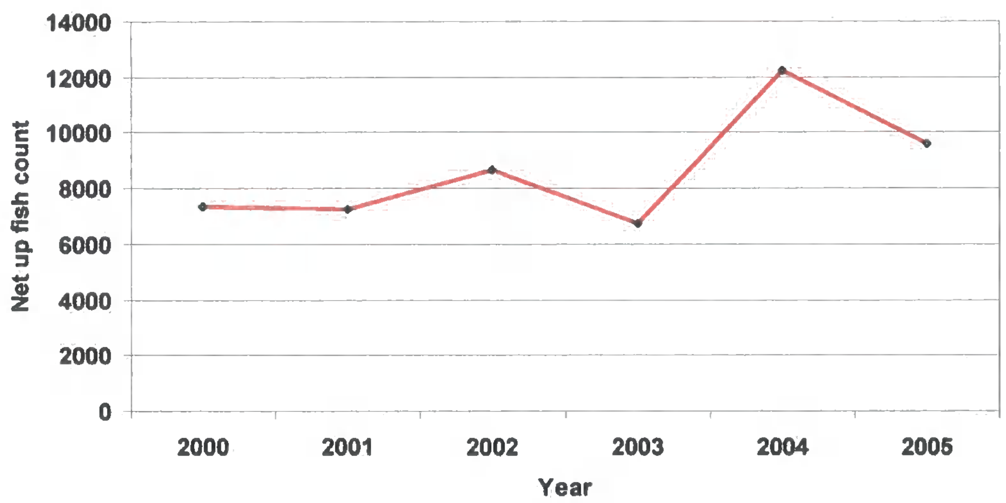


Figure 3.6: *Number of salmon returning to the Eden catchment (2000-2005), as recorded by the Environment Agency's resistivity fish counter located at Corby on Eden.*

Between 1999 and 2002 the Environment Agency also undertook a radio-tagging research programme to examine the spatial distribution of Atlantic salmon spawning in relation to sea-age and the timing of return migrations, and to assess the survival rate of fish caught and released by anglers. 302 salmon were tagged in total, of which 184 were successfully tracked to their spawning location. Clear spatial stratification of spawning was found in relation to sea-age and run time. Multi-sea-winter, spring-run fish were primarily found in the River Lowther and River Eamont, with a smaller number tracked to Hilton Beck, River Caldew, River Irthing, River Lyvennet and the main-stem Eden. Grilse and summer/autumn run fish were primarily observed to spawn in the main-stem. The study also concluded that over 85% of rod caught salmon can be expected to survive to spawning after being caught and released by anglers (Gowans, 2004).

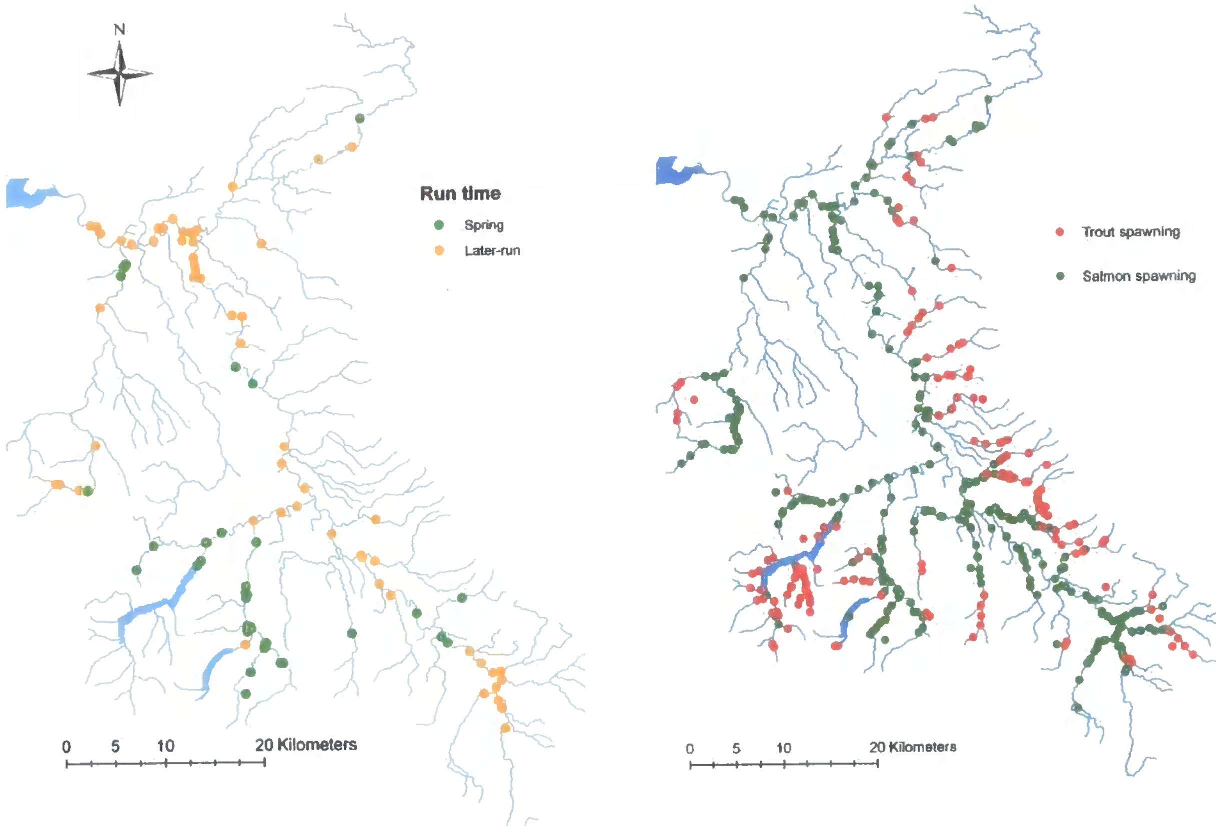


Figure 3.7: (a) *Spawning locations of Atlantic salmon stratified by migratory run-time. Based on data from the Environment Agency's radio-tagging programme for 1999 and 2000 (Gowans, 2004). (b) Prime spawning locations for salmon and trout within the Eden catchment based on radio-tagged data (Gowans, 2004) and electrofishing data (Source: Eden Rivers Trust)*

The dataset on radio-tagged spawning locations has been combined with electrofishing data from the Eden Rivers Trust for 2002-2006 to produce a map of prime spawning locations for salmon and trout within the Eden catchment (Figure 3.8). It has been assumed that reaches of excellent or good fry numbers correspond to reaches with prime spawning habitat within the immediate vicinity.

3.4 Aerial photography

High-resolution aerial imagery was identified as a major data requirement for the assessment of several riparian and in-stream habitat variables (Table 3.1). Traditionally, this information would have been collected via a ground reconnaissance survey based on geomorphological survey techniques (Table 1.4). These techniques are particularly suited to providing detailed data at small spatial scales, but are considered prohibitively time consuming and costly at the catchment

(>1000 km^2) scale. For example, APEM Ltd estimate that using their walkover survey technique, only 4km per day can be surveyed by a single surveyor, plus an additional laboratory-based day is required to covert hand drawn maps into GIS format (*pers. comm.* K. Hendry). The Environment Agency's River Habitat Survey (RHS) is estimated to take 1 hour per 500m site at a cost of £40-120 per site, plus an additional £20 for data entry and quality checking (*pers.comm.* M. Diamond). Recent technological advances have improved the efficiency of these techniques. For example, portable GPS equipment now enables accurate and rapid recording of feature locations and physical measurements within the field, facilitating GIS integration and quantitative assessment on retum to the laboratory. This has led to the development of survey methodologies such as GeoRHS (Newson and Hill, 2004), whilst developments such as handheld GIS mapping equipment (e.g. Gismo) can also reduce the time spent inputting hand drawn maps into electronic format. Survey methodologies can be modified to reduce time and costs (Hendry and Cragg-Hine, 1997), but this typically represents a trade-off between speed, and accuracy and coverage. There are a number of possible modifications. First, it may be possible to reduce the length of river surveyed by employing a suitable sampling strategy. For example, RHS surveys sample a number of 500m reaches, and within each reach 10 equidistant spot-check sites are surveyed in detail. Care must be taken to ensure the spatial resolution of sampling captures the heterogeneity of habitat present. However, such an approach is not considered suitable for this research which requires continuous data. Second, it may be possible to reduce the amount of data collected. Surveys may be carefully designed for a specific purpose and only relevant data collected. This is the aim of user-specific surveys such as APEM's rapid salmonid habitat survey, which only collects data relevant to salmonids. However, should further data be required at a later date, a new survey will be required increasing costs. The amount of data collected may also be reduced by visually estimating the extent and occurrence of features such as substrate size or percentage riffle rather than accurately measuring them. However, this may introduce subjective error and relies on the expertise of the individual surveyor. Third, it may be possible to prioritise areas for evaluation. With existing knowledge of general habitat condition (e.g. riparian land use) and fisheries performance, areas may be prioritised according to their likely need for habitat improvement. However, such an approach may introduce subjective bias according to the factor that is perceived to be limiting. This approach is not considered appropriate for this research which requires objective, catchment-wide coverage.

Alternatively, the use of high-resolution remote sensing is increasingly advocated to capture river related features as demonstrated by Table 1.5. The advantages of using remote sensing are as follows:

(1) it facilitates rapid coverage of an extensive area thereby enabling catchment-wide and continuous data to be acquired in a cost effective manner;

(2) it enables data collection in areas otherwise inaccessible due to access constraints or difficult terrain, thereby providing a continuous data source;

(3) it provides a permanent visual record of the riparian corridor that can be revisited to extract additional information or gather further expert opinion at a later date;

(4) as a permanent record, it can be used to monitor environmental change over time;

(5) imagery provides a powerful tool for explaining environmental issues to the public, helping to encourage co-operation from local communities, landowners, and funders;

(6) it can be collected directly into a digital format removing the need for data entry upon returning from the field; and

(7) automatic classification procedures and GIS analysis allow accurate assessment and quantification of features reducing subjectivity when compared with traditional walkover surveys.

A vast number of remote sensing platforms and sensors exist from satellite to airborne collecting data across a range of spectral, spatial, and temporal resolutions. Increasing resolution increases the ability to identify and classify smaller scale features. However, increased resolution comes at increased cost both in terms of data capture and in terms of computational processing and data storage requirements. Therefore, a balance must be achieved between the resolution required and cost. In selecting imagery for use in this research, it is the spatial and spectral resolutions that are important to consider. The spatial resolution required is largely determined by the physical dimensions of the river and size of habitat features being classified. A higher stream width to pixel resolution ratio will result in greater accuracy due to the reduced impact of pixel mixing (Legleiter *et al.*, 2002). This is especially important when distinguishing between various water features such as riffles and glides where pixel unmixing is not viable due to similarities in reflectance (Marcus *et al.*, 2003). Similarly, it is important for identifying bank features such as erosion which may occur within a very narrow zone of 1-2m between the channel and wider riparian zone. A resolution of approximately 20cm has been selected and is considered acceptable to classify habitats in streams down to 2m (10 pixels) wide. This should enable feature variation across the channel to remain detectable and facilitates considerable coverage of important spawning and juvenile tributaries in the Eden catchment to be obtained. The spatial resolution achieved is determined by the focal length of the sensor used and its height above the ground surface. Thus, high spatial resolution data are most typically delivered by airborne platforms (e.g. helicopters and planes). The spatial resolution achieved by space-borne sensors is increasing (e.g. IKONOS – 1m panchromatic or 4m multi-spectral; QuickBird – 5m) but these are not yet comparable with that achieved by airborne sensors.

In terms of spectral resolution, a number of papers have demonstrated the use of multi-spectral (Puestow *et al.*, 2001) and hyper-spectral (Marcus, 2002) imagery for mapping in-stream salmonid habitat. However, due to cost at the current time, such imagery typically remains in the research domain, or where used by government agencies it is limited in its spatial distribution. Instead, true colour photography has been selected for this research, which due to recent advances in digital camera technology (increased pixel resolution) can now be provided rapidly and relatively cheaply using plane mounted cameras.

3.4.1 Aerial image capture

A true colour digital aerial photography survey covering approximately 660km of the River Eden and its tributaries was commissioned by the Eden Rivers Trust and undertaken by Compass Informatics during 2004. The survey was solely focused on the river channel and riparian zone. This is the first time such a survey has been undertaken at this scale in the UK. The image capture system developed by Compass Informatics comprised a 12 Mega pixel digital camera mounted on a simple metal platform attached to the floor of a Cessna 172 aircraft. The camera was able to slide along this mount into the cavity space of a modified rear luggage door allowing a near vertical ground view (Figure 3.8). The camera was triggered at fixed intervals by electronic signals sent from a laptop computer, which was also connected to a GPS enabling the nominal centre point for each triggered image to be recorded (*pers comm.* G O'Riain). Images were instantaneously downloaded to the laptop and stored in TIFF format. By flying approximately 900 metres above the ground surface near-vertical imagery with a ground resolution of 20cm was collected. Each image covered approximately 200m either side of the channel and 600m in a straight-line downstream direction. Overlap of approximately 30-60% was achieved between sequential images.

Figure 3.8: *Digital camera system used by Compass Informatics to collect aerial imagery of the Eden catchment.*

Using GPS technology the plane followed pre-defined flight paths along the channel of selected tributaries. Tributaries were selected in ArcGIS using the Ordnance Survey (OS) Landline and 1:50,000 Raster datasets provided by the Environment Agency under sub licences (Reference: Environment Agency, 100026380), and a digitised channel centre-line of the Eden catchment produced under the CHASM (Catchment Hydrology and Sustainable Management) project (www.ncl.ac.uk). Only tributaries >2m wide were selected as identified from OS Landline data. Streams in excess of 2m width are shown by a double blue line within the LandLine dataset, those less than 2m width by a single line. Due to cost restrictions images were not collected for the main stem Eden downstream of the River Eamont confluence as this was considered to represent less suitable juvenile salmonid habitat. During the flight, a navigation system developed by Compass Informatics was used to locate selected flight paths. First, a moving map connected to the GPS was used to locate selected streams and second, images from a video camera mounted next to the digital camera were used to ensure position directly above the channel was maintained. It was not always possible for the plane to follow the exact path of the river (e.g. around a meander bend) and in such circumstances, using the video information, the pilot would repeat fly the section until it was considered that adequate coverage had been achieved. However, due to the difficulties involved in this some small gaps within the image sequence did occur.

It was important to ensure the survey was conducted under the right environmental conditions to maximise image quality and visibility of the required features. The condition required were:

(1) a dry period of at least several days preceding the survey flight in order to ensure low river flows;

(2) clear visibility with either no cloud or high cloud during the survey to ensure collection of sharp, clear imagery;

(3) limited tree and aquatic macrophyte foliage to maximise visibility of channel geomorphology; and;

(4) high sun angle to minimise the impact of shadows.

25cm resolution aerial photography is commercially available with national coverage from Getmapping plc (www1.getmapping.com). However, after evaluation, it was not considered suitable for use within this research as collection under correct environmental conditions (primarily low flow) could not be guaranteed resulting in reduced visibility of channel geomorphology. Even with a purpose designed survey, meeting these conditions proved challenging. The weather in the Eden catchment is highly changeable and unpredictable making assessment of conditions and agreeing activation of the survey team based in Ireland somewhat problematic. Surveying was further restricted by a military, low-fly training zone which covers the majority of the catchment and which is active most week days. However, the survey was completed with 3 activations of the survey team between February 21-25, April 13-14 and June 14-15 2004. Figure 3.9 presents the coverage achieved. Unfortunately, the survey team were unable to capture images of the River Leith during any activation due to weather conditions.

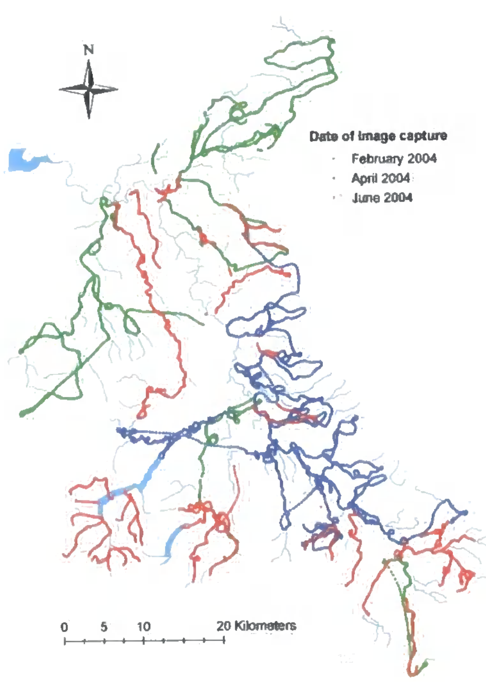


Figure 3.9: *Aerial photography coverage achieved for the Eden catchment during 2004.*

Collection of the data during three different months has enabled assessment of the image quality achieved at three different times of year. A sample of images, one from each activation, is shown in Figure 3.10. Images collected during February are considered to be of the lowest quality. It was initially thought that February might be a good time to collect imagery due to the lack of leaf foliage and aquatic macrophytes. However, short days and low sun angles at this time of year increased the impact of shadow, especially where riparian trees were present. This eliminated the advantage of reduced foliage. Heavy morning frosts also reduced the visibility of riparian land cover, whilst further complications were encountered as freezing temperatures led to shutdown of the camera equipment. Images captured during April and June are considered to be of much higher quality, as the impact of shadow is reduced. Those collected in April were captured prior to full leaf foliage and therefore represent maximum visibility of channel geomorphology, whilst those captured in June do have restricted channel visibility in heavily wooded reaches due to increased foliage. Following this comparison it is recommended that any future surveys of the Eden, and similar catchments, be undertaken between late March and early May.

Figure 3.10: *Comparison of aerial photographs captured in (a) February, (b) April and (c) June, 2004.*

3.4.2 Image rectification

In order to meet point (3) of the data requirements (Section 3.2), it was essential that the aerial imagery be accurately rectified (geo-refrenced). This involves alignment of an image to a map so that the image is planimetric, (Schowengerdt, 1997 p329). This not only involves correctly orientating the image in space but also correcting for geometric distortions present in the raw imagery as a result of variations in flying height, velocity, topography and panoramic distortion (Lillesand and Kiefer, 1994). This process was completed by Compass Informatics.

A number of common 2-D rectification procedures exist typically based upon polynomial distortion functions or transformations. These techniques are valid in regions of relatively constant terrain. However, as discussed in Section 1.3, the topography of the Eden catchment is highly variable and complex in many areas, with elevations ranging from 0 – 950m above sea level. If not accounted for in the rectification procedure, such complex topography can lead to severe distortion in the final rectified image due to variations in the Z-field. To deal with this issue 3-D procedures for ortho-rectifying imagery have been developed. These project the image over a digital elevation model (DEM) so that every pixel appears to be viewing the earth from directly above.

The traditional approach to rectification is to manually locate ground control points (GCPs) on the imagery and associate these with known X, Y, co-ordinates taken from maps such as digital OS data or collected in the field using visible makers and GPS. GCPs should be evenly distributed across the image, of high contrast in all images of interest, of small feature size and unchanging over time (Schowengerdt, 1997 p335). As such, readily identifiable and stationary points such as crossroads and building edges are typically selected. In relation to this research there are two major drawbacks to this approach. Firstly, manual ground control point location is very time consuming and therefore costly, especially for situations such as this where there are a high volume of images (1000+) to process. Secondly, whilst identifying ground control points in a built environment is relatively easy, in rural and remote environments such as the Eden catchment, this is much more problematic. In such environments, field boundaries and rivers are often the only identifiable features, but both of these features are typically dynamic over time.

Instead the approach adopted by this research, as identified by Compass Informatics, was to use a piece of new software, EnsoMOSAIC developed by StoraEnso, which facilitates automatic pattern recognition between adjacent images to identify hundreds of tie points (equivalent to ground control points). This ability enables a large number of images to be processed rapidly, the developers claim that one standard PC can process 200-1000 images in 24 hours (StoraEnso, 2006), and it does not rely on built features. To operate, the software requires that the camera position (recorded by Compass Informatics using GPS within the aircraft) for each image is known, together with camera parameters such as focal length and principal point. Additionally, the greater the percentage overlap between images, the greater the ability for the software to identify tie points. Compass Informatics typically achieved 30-60% overlap between images in both the lateral and longitudinal plane. Images to be included in the rectification procedure were selected and ordered into a suite of longitudinal sequences (typically 1-10 images) by an operator at Compass Informatics.

As described by StoraEnso, (2006) the image mosaic procedure comprises four stages.

1. **Image orientation:** The initial orientation of the images is calculated either automatically by utilizing the camera attitude parameters or manually by defining links between the images.

2. **Tie point location:** Tie points between images are located automatically using image correlation. These can be manually edited after selection and additional ground points added if required to improve the accuracy of image rectification.

3. **Image rectification:** Bundle Block Adjustment (BBA) was applied for automatic image rectification using the GPS recorded camera positions collected during image capture. Images were rectified to OS National Grid of Great Britain. At the same time using stereo-imagery capability, a Digital Elevation Model (DEM) was created. Altitude values are typically calculated for 20-50 points per image, but up to 1000 points can be used if required. Obviously the greater the number of points selected the longer the processing time, but more accurate the DEM. The DEM is then used to generate ortho-rectified imagery.

4. **Mosaic creation:** The rectified images were then joined together to create a final mosaic. Output was in TIFF file format with a pixel resolution of 20cm. The software does offer a number of colour adjustment options such as histogram equalisation, to standardise for variations in illumination, reflectance properties of surface materials, cloud and topographic shadow between sequential images. This improves visualisation by creating a seamless mosaic. However, for the purposes of this research, specifically the application of aerial imagery to water depth assessment, it was decided not to apply any adjustment factors as

the raw reflectance values were required. For further discussion of this issue see Chapter Four (Section 4.3.3).

Breaks between mosaics occurred or were manually inserted where there was a gap in the imagery captured; where the overlap between two adjacent images was not sufficient to facilitate pattern matching; or where the output file size would be too large to handle efficiently within a standard PC. Following provision of the imagery by Compass Informatics each mosaic was screened to assess its quality and any which failed the quality assessment were returned for additional processing. Two main reasons for image rejection were identified:

(1) Poor spatial rectification: If the geographic location of a mosaic or specific features (particularly) the river within a mosaic deviated significantly (>50m) from OS LandLine data or between sequential images the mosaic was rejected. This was corrected by manually increasing the number of GCP's particularly around the river.

(2) Topographic distortion: This primarily occurred in areas of complex topography and was corrected by increasing the number of altitude points used to create the DEM and its output resolution.

Increased numbers of GCPs and altitude points could have been used in the initial rectifications to avoid the above issues but this would have increased computational time and expense for many mosaics where the added detail was not necessary. Approximately 15% of the mosaics were returned for additional processing. Final mosaics were stored in TIFF format on an external hard drive, categorised by sub-catchment, and tagged with the date of acquisition and mosaic number (e.g. 20040614_03_01).

3.5 Digital Terrain Model (DTM)

As identified in Table 3.1, a high-resolution DTM is required to calculate channel gradient, infer in-stream flow type and derive topographic attributes required for environmental modelling (Chapter Five) such as, valley slope, the topographic index, flow routing and ultimately to evaluate catchment-channel hydrological connectivity. Digital terrain models are 3-D representations of the Earth's bare surface. Features such as vegetation, buildings and cultural objects are removed to leave just the underlying surface. This is slightly different to a digital elevation model (DEM) which includes buildings and trees. This is an important distinction to make when the model is being specifically applied to the assessment of natural terrain and not manmade features. A number of

DTMs are available commercially within the UK (Table 3.6), and it is important to consider their resolution, accuracy, availability and cost in determining which one to use in this thesis.

Table 3.6: *Resolution of Digital Terrain Models available for the Eden catchment*

DTM	Horizontal resolution (m)	Vertical data quality (m)	Catchment-wide coverage
OS Panorama®	50	±5	Yes
NEXTMap Great Britain™	5	±1	Yes
LiDAR	~1-2	~±0.25	No

3.5.1 Effects of DTM resolution

DTMs are widely used within geomorphological and hydrological modelling studies to derive a variety of topographic attributes. However, the numerical value of parameters derived from DTMs has been observed to vary considerably with DTM resolution (grid cell size) due to differences in the representation of the landscape at different scales (Schoorl *et al.*, 2000). These effects are well documented within the scientific literature and variables shown to be sensitive to DTM resolution include, valley slope, upslope contributing area, topographic index, and flow path location (Zhang and Montgomery, 1994; Brasington and Richards, 1998; McMaster, 2002). This in turn impacts upon the spatial distribution of runoff processes and associated sediment transport affecting model hydrological and geomorphological predictions. A coarsening of resolution typically results in reduced predictive capability. For example, Zhang and Montgomery (1994) analysed the cumulative frequency distributions of slope, upslope contributing area and topographic index derived from 5 DTMs ranging from 2-90m resolution. Both the mean and local values of all three variables were sensitive to DTM resolution, particularly in areas of steep topography. At coarser resolutions, slope was observed to decline, whilst the upslope contributing area and topographic index increased. Schoorl *et al.* (2002) evaluated the impact of 5 DTM resolutions between 1 and 81m on erosion and sedimentation rates predicted by a simple sediment transport model, concluding that coarser DTMs overestimated erosion and underestimated resedimentation. At coarser resolutions, many topographic features such as hollows and low-order channels may not be resolved. But the question still remains as to where the critical resolution lies? A number of people have commented that the resolution must be finer than the average hillslope length identifiable in the field, enabling capture of variability in hillslope morphology. Resolutions of the order 10 to 150m have been proposed depending on the catchment of study (Zhang and Montgomery, 1994; Brasington and Richards, 1998; McMaster, 2002). However, more recent evidence suggests that the processes controlling the spatial distribution of runoff and material transport may be conditioned by local, often sub-field scale, hydrology occurring at spatial scales in the order of <10m (Western *et al.*, 1999; Lane *et al.*, 2004; Heathwaite *et al.*, 2005), often related to quite subtle topographic attributes (Lane *et al.*, 2006). Such resolutions must be captured in the original data as decreasing grid size beyond the resolution of the original survey data does not increase the accuracy of the land surface representation. Conversely, it potentially introduces interpolation errors (Zhang and Montgomery, 1994). Additionally, it is not only horizontal resolution that is important to consider, vertical precision has also been shown to influence the derivation of topographic parameters and hydrological predictions (Kenward *et al.*, 2000), particularly in low-relief landscapes, where small scale terrain features such as roads, ditches and irrigation channels can significantly affect runoff flow paths (Duke *et al.*, 2006). Based on the above, it is considered that the OS Panorama 50m DEM is not adequate for use within this research. Studies using this DEM to calculate channel gradient have reached the same conclusion (Coley, 2003). Of the remaining models, LiDAR (Light Detection And Ranging) offers the finest spatial resolution and highest vertical precision. However, it is currently unavailable for the entire Eden catchment, having only been captured within urban and coastal areas for flood defence purposes by the Environment Agency. As discussed below LiDAR is also relatively costly with added detail resulting in increased computational demands. This leaves the NEXTMap Great Britain ™ DTM which is available for the entire Eden catchment at relatively low cost and with a relatively high spatial resolution. As such it is the DTM selected for use within this thesis.

3.5.2 NEXTMap Great Britain ™

The NEXTMap Great Britain ™ DTM has been developed by Intermap Technologies Inc from survey data collected during 2002 and 2003. It was commissioned by Norwich Union to support their FLOODMAP product and is now commercially available for the whole of England, Scotland and Wales. NEXTMap was produced using interferometric synthetic aperture radar (IFSAR) technology and has a horizontal resolution of 5m, with a vertical precision of ±1 m, as reported by Intermap (2003). Independent studies have estimated the data to be more precise than this (to ±0.9m vertical precision) in similar upland catchments (Reid *et al.* in review). It is projected to the OSGB36 Grid.

IFSAR is a relatively new digital mapping technology developed for US military applications (Sanders *et al.*, 2005). Images are created by combining signals received from two side-looking radar antennae mounted on an aircraft and displaced by a known distance (1m for NEXTMap). X-band radar with a radar pulse wavelength of 3cm is used. One antenna acts as both transmitter and receiver, the second as a receiver only. Due to the separation of the antennae, radiation reflected from a point on the ground will strike each antenna at a slightly different moment in time. This is known as the phase difference, and coupled with precise aircraft positional data, provides the information required to measure the elevation points (Intermap, 2003; Sanders *et al.*, 2005). Terrain information is then generated from the raw elevation data by digitally removing vegetation, buildings and other cultural features. This is achieved using TerrainFit® software which derives the terrain surface by interpolating measurements taken from bare ground. IFSAR is currently considered the most economic airborne technology for collecting high-resolution topographic data. LiDAR does collect data with a higher resolution and vertical precision but this is achieved at increased cost. IFSAR may be collected from relatively high flying heights, 8,500 metres compared with a typically flying height of 1,500 metres for LiDAR (Smith *et al.*, 2006). Computational processing requirements are also lower, facilitating use within catchment-scale environmental models. Radar technology is also independent of sunlight for viewing and can therefore be collected during day or night. It is also reasonably weather independent permitting data capture during rainy and cloudy conditions (Sanders *et al.*, 2005). However, a number of errors have been reported. For example, NEXTMap has been reported to be unreliable in wooded areas as it is unable to penetrate vegetation canopies (Smith *et al.*, 2006). This is acknowledged by the developers who state that in wooded or built areas greater then $100m^2$ TerrainFit® may falsely sample local minima within the canopy that are not necessarily on the ground. This creates 'edge effects' near the boundaries of such areas where interpolation between true ground and falsely elevated points creates intermediate elevations. The transition zone is typically less than 25m horizontal (Intermap, 2003). Care should therefore be taken when evaluating results in such areas. Other potential sources of error noted by Intermap (2003) are: (1) in areas of steep slopes, 20-30^{o} the vertical error may double; (2) rapidly changing terrain features such as embankments may not be preserved due to inadequate sampling density; and (3) due to the side-looking technology of the IFSAR sensor, layover may occur in front of tall structures with shadow behind (Figure 3.11). Layover is an effect of object height, where the top of the object is illuminated before the base thereby obscuring part of the 'ground range'. It is particularly prevalent in mountainous regions. Shadow occurs in regions which cannot be reached by the radar pulse. The former may be mitigated by post-processing whilst the later produces a region of no data. This may be in-filled with data from an adjacent pass during image merging.

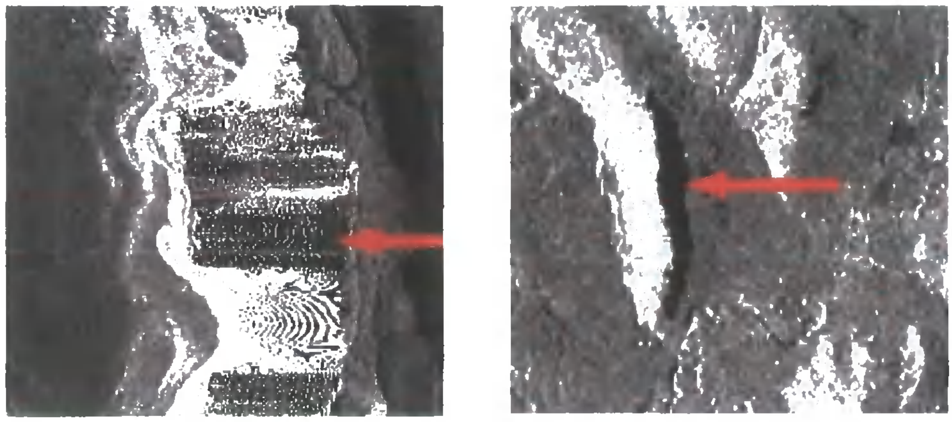


Figure 3.11: *The visual effect of (a) layover and (b) shadow on IFSAR data (reproduced from Intermap, 2003).*

Visual inspection of the relief shaded NEXTMap DTM for the Eden catchment (Figure 3.12), has indicated that such effects appear to be minimal. The catchment is not heavily wooded, except in the Northern Irthing sub-catchment, and therefore the impacts of woodland are also considered to be minimal. The NEXTMap Great Britain ™ dataset has been successfully used by a number other projects to undertake flood risk mapping (Sanders *et al.*, 2005) and to map glacial landforms (Smith *et al.*, 2006). The NEXTMap DTM for the Eden catchment was acquired from Getmapping plc by Eden Rivers Trust under a perpetual charitable licence at cost of £1 per km^2. The data were provided as 100km^2 tiles in ASCII X,Y,Z format and converted to ESRI GRID (raster) files using ArcGIS. The individual raster files were then merged into a single seamless DTM for the catchment.

Figure 3.12: *Relief shaded NEXTMap Great Britain ™ DTM for the Eden catchment*

3.6 Land cover data

Information regarding catchment-wide land use is required to investigate the relationship between salmonid populations and the spatial distribution of land use, classifying the landscape according to its risk of pollution production based upon the premise that certain land uses and land management activities are associated with a greater risk of pollution and fine sediment production. It is generally considered that the more intensive the land use, the higher the associated risk (e.g. Robinson, 1999; Caruso, 2001). As with the previous data sources, it is again important to consider the spatial resolution required. Land use and land management decisions are typically made at the individual farm or even field scale, and this is therefore the scale at which data are required. Such data are collected through the Agricultural Census undertaken by the UK government Department for Environment, Food and Rural Affairs (DEFRA). This is publicly available at the ward level (e.g. through Edina). However, data at the farm scale remains confidential. At the ward level land uses are averaged across a large area (in the order of 10-100km^2). This spatial resolution is too coarse for this research as it fails to capture the detailed spatial distribution and mosaic which actually occurs within the landscape. As discussed in Section 3.5.1, the spatial distribution of hydrological runoff and material transport processes may be occurring at a sub-field (<10m) scale (Lane *et al.*, 2006). Unless these processes can be associated with land use and land management at a similar scale, accurate conclusions cannot be drawn about the impact of land management upon the in-stream environment and ultimately upon salmonids. Alternatively, remote sensing can be used to gather high-resolution spatial information on land cover, at a catchment-scale. Whilst this does not provide detailed information as to the specific management strategies applied at a particular point in the landscape, or variation in land use over time, it can be used as a proxy, as certain practices are more likely to be associated with or result in particular land covers. For example, by its very nature improved pasture is more likely to have received fertiliser, slurry or manure applications than unimproved pasture and therefore to be a source of nutrients such as nitrate and phosphate. Similarly, improved pasture is more likely to be intensively stocked and heavily grazed than unimproved pasture due to the increased nutritional value of grass. Consequently these areas are likely to be at greater risk of erosion due to reduced vegetation cover and increased soil compaction than unimproved, more extensively stocked areas (Owens *et al.*, 1997). However, some degree of caution should be used as land cover cannot always infer land use (Fuller *et al.*, 2005). For example, improved pasture may not be actively grazed, instead being used to provide cut silage. Despite such uncertainties, classification of land cover from remotely sensed imagery offers the best indication of land use at the current time. Within the UK a national database of land cover at a spatial resolution of 30m (sub-field scale) has been developed by the Centre for Ecology and Hydrology (CEH) and this is the data used in this thesis.

3.6.1 UK Land Cover Map 2000 (LCM2000)

Land Cover Map 2000 (LCM2000) is a comprehensive survey of UK broad habitats giving vector digital maps from segment based classification of remotely sensed satellite data (Fuller *et al.*, 2005). It was developed using satellite imagery from Landsat Thematic Mapper (TM) which has a spatial resolution of 30m, collected during 1998 and 1999. A combination of winter and summer imagery from the red, near infrared and middle infrared spectral bands was used, enabling discrimination between bare and developed land, annual cropping and deciduous and evergreen vegetation. Automated image processing techniques using supervised classification and vector segmentation, were applied to the imagery to identify 16 Target classes mapped as vector polygons. These are further subdivided into 26 Subclasses (Level 2 vector dataset) and 72 Variants (Level 3 vector dataset), wherever image quality made this possible (Fuller *et al.*, 2002). Appendix 2 details the classification system used which is based upon the 'broad habitat' classification developed for reporting under the UK Biodiversity Action Plan. A comprehensive description of the image processing methodology is provided by Fuller *et al.* (2002). The developers comment that a direct evaluation of LCM2000 precision is not yet possible, but map accuracy is estimated to be in the order of 80-85% for broad habitat classes increasing to approximately 90% for the 16 Target classes (Fuller *et al.*, 2005). These accuracy estimates refer to the year of data capture, now 8 years ago, and it is probable that the dataset may misrepresent the actual nature of land management. However, as farm level data from the Agricultural Census remains confidential, this was judged the best alternative. The LCM2000 Level 3 vector data for the Eden catchment were freely sub-licensed to Eden Rivers Trust by the Environment Agency and provided as ESRI ArcView polygon shapefiles. This has been converted into a raster (ESRI ArcView GRID) dataset with a pixel resolution of 20m based on the Level 2 subclasses, to facilitate environmental modelling. The Level 2 data were selected representing a compromise between reliability (overall accuracy of 80-85% at the time of capture) and detail (Figure 1.6(d)).

3.7 Rainfall data

The final data requirement for environmental modelling of catchment-scale controls is a catchment-wide layer of time integrated rainfall. This is required to calculate the dilution potential of runoff on concentrations of material transported from the catchment to the channel. The concept is simple: the higher the ratio of water to material, the lower the concentration of pollutants and therefore the lower the risk to aquatic ecology. Dilution potential relates to the volume of water draining to a point in the channel network and is a function of the upslope contributing area and the amount of rainfall received. The spatial distribution of rainfall is highly variable at a short temporal scale related to individual storm tracks. However, over longer timescales a more stable pattern in the spatial distribution is observed, related to geographic and topographic factors such as location, terrain height and shape, and urban and coastal effects (Perry and Hollis, 2005). This research is concerned with investigating spatial distributions and therefore requires a long-term, time integrated, dataset of spatial rainfall distribution. The specific data used are the 1961-2000 baseline, 5km resolution, gridded dataset of mean annual precipitation sourced from the UK Meteorological Office and the UK Climate Impacts Programme 2000 (UKCIP2000). A two-stage process was used to create this national dataset. Firstly, multiple regression analysis of precipitation with a range of geographic and topographic factors such as easting and northing, terrain elevation, and percentage open water was undertaken, followed by inverse distance-weighted interpolation of the model residuals. The regression surface and the interpolated residual surface were then added together to get the final gridded datasets. (Perry and Hollis, 2005 – provides a comprehensive description of the dataset development). Data were provided as an ESRI ArcView GRID (Figure 1.6(c)).

3.8 Impassable barrier survey

As identified in Chapter Two (Section 2.5.1), impassable barriers to salmonid migration can have a significant impact upon the spatial distribution of salmonid spawning densities and the utilisation of spawning habitat. The level of their impact is determined by their location within the catchment and the amount of habitat they render inaccessible and therefore unusable. To provide information on this, the precise location of barriers (both natural and manmade) is required. These data have been provided by the Environment Agency as detailed in the River Eden Catchment Salmon Action Plan (2000). A walkover survey of the catchment was undertaken to locate and assess potential barriers. GPS coordinates of barrier location were recorded, together with basic height and width parameters. The classification of a barrier as potentially impassable was initially based upon expert judgement and comparison with electrofishing data collected above the obstruction. The absence of salmon was taken as an indication that the barrier was impassable. The artificial barriers classified as potentially impassable have more recently been subjected to a professional and detailed survey by a fish pass consultant (Beach, 2006). The judgement as to whether a barrier is impassable is primarily based upon the measurement of the hydrometric head across each weir or falls. A site is not considered to present an obstruction if the head difference across it is less than 0.45m for salmonids, or less than 0.3m for coarse fish. The depth below a structure should also be at least 0.5m, but conditions are made difficult for upstream migration if the downstream face of a weir consists of a long slope that is sharply truncated but not inundated (Beach, 2006). Data on the location of impassable barriers has been provided by the Environment Agency and Eden Rivers Trust as a Microsoft Excel spreadsheet detailing XY coordinates. This has been converted to an ESRI ArcGIS point shapefile. Figure 3.13 illustrates two of the impassable barriers located in the Eden catchment.

(a)

(b)

Figure 3.13: *Impassable barriers to salmonid migration found in the Eden catchment, (a) Swindale Beck Upper Weir and Cam Beck Pipe Bridge. Photographs courtesy of Eden Rivers Trust.*

3.9 Summary of data requirements and validation

This thesis is heavily reliant on the ability to capture data regarding salmonid habitat controls at a range of scales, together with catchment-wide data on salmonid abundance. The ability to acquire such data has been a major obstacle to this type of research in the past, as traditional data collection techniques have focused on collecting detailed information at small spatial scales. However, recent technological advancements in remote sensing, GIS, GPS, ecological surveying techniques, environmental modelling, and the development of many readily available regional and national digital datasets are increasing the possibility of achieving this. The aim of this chapter was to identify the major data requirements of this research and consider the data sources available for the Eden catchment. A suite of criteria including catchment-wide coverage, spatial resolution, spatial referencing, digital format, continuous coverage, current data and cost has been considered in selecting the final datasets which are presented in Table 3.7.

Table 3.7: *Datasets for the Eden catchment.*

Data requirement	Dataset	Developer	Supplier
Digital Map data	OS Land-Line OS 1:50,000 raster	Ordnance Survey	Environment Agency sub-licence
River centre-line	River centre-line digitised from OS data	CHASM, Newcastle University	CHASM, Newcastle University
Aerial Photographs	20cm true colour digital photographs	Compass Informatics	Compass Informatics
Digital Terrain Model	NEXTMap Great Britain DTM	Intermap Technologies Inc	Getmapping plc
Salmonid distribution and abundance	Semi-quantitative electrofishing data	Eden Rivers Trust	Eden Rivers Trust
Land Cover Data	Land Cover Map 2000	Centre for Ecology & Hydrology	Environment Agency sub-licence
Rainfall data	5km resolution precipitation data 1961-2000	UK Climate Impacts Programme	UK Met Office
Impassable barriers	Barrier XY co-ordinates	Environment Agency	Environment Agency

As Table 3.7 indicates this research relies heavily on data sourced from external organisations. As such, evaluation of data accuracy and the uncertainty introduced by selecting these sources must be undertaken. Many of the data sources selected have been subjected to intense validation and accuracy assessment procedures by the data developers or other users as discussed throughout the chapter. Within this thesis it is proposed that the performance of these datasets will be further validated through: (1) their ability to derive secondary habitat variables accurately; and (2) their ability to explain variance in the salmonid populations monitored through the electrofishing programme. To this end, Figure 3.14 illustrates how the various data sources will be integrated, processed, analysed and validated throughout this thesis. Discussion of these processes forms the main focus of Chapters Four (aerial imagery analysis) and Five (catchment-scale environmental modelling).

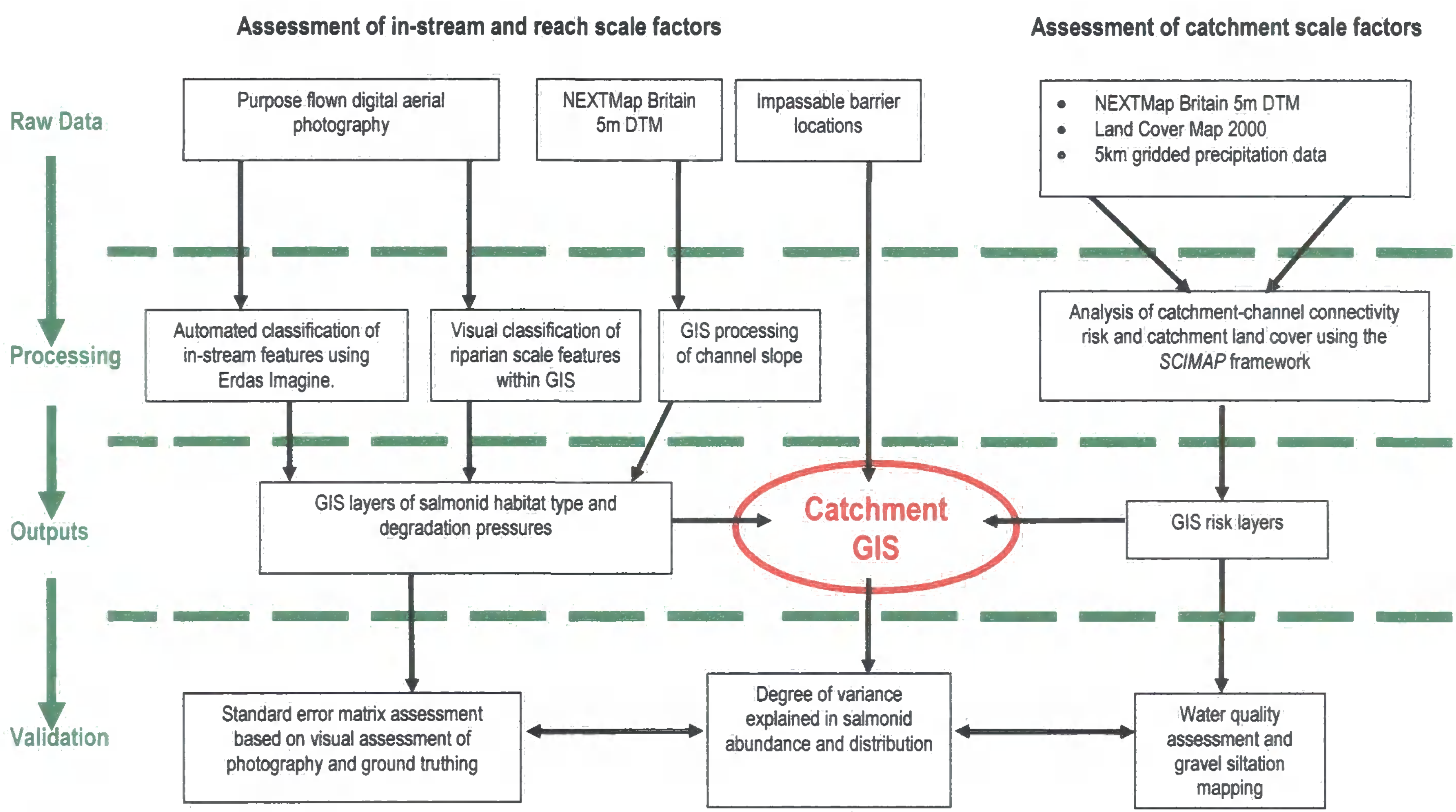


Figure 3.14: *Diagrammatic representation of data integration, processing, analysis and validation throughout the thesis*

Chapter Four - Riparian and in-stream habitat assessment using remotely sensed data and GIS

4.1 Introduction

Chapter Three introduced the broad-scale datasets selected for use within this research. As discussed, the real power of these datasets becomes apparent when they are combined with environmental models, or subjected to advanced GIS and image processing techniques. It is this secondary processing which forms the basis of Objective 2 of this thesis: *To employ recent advances in remote sensing, GIS and environmental modelling, to identify, to develop and to validate tools for quantifying salmonid habitat at the catchment-scale, appropriate to each habitat control and scale of control.* With regard to delivering Objective 2, the aim of this chapter is to identify and develop tools for the derivation of salmonid relevant habitat data at the- stream and riparian-scale based on the data sources described in Chapter Three, specifically 20cm digital aerial photography and the NEXTMap Great Britain ™ 5m digital terrain model (DTM).

In terms of developing tools for environmental analysis using aerial photography and digital terrain models, three main approaches identified within the scientific literature are applied here: (1) visual assessment of imagery, and mapping of attributes using vector GIS capabilities; (2) raster-based image processing to analyse and map features based on their reflectance properties; and (3) raster-based GIS processing of digital elevation data. Methodologies, validation, and evaluation of each of the three tools will be presented in turn. Validation and evaluation of each tool's capability and applicability to this research will be achieved by first, assessing habitat classification accuracy through the use of ground validation and established accuracy assessment procedures based on the error matrix (Congalton and Green, 1999) and second, and equally importantly, by assessing their practicality and cost compared with traditional walkover survey techniques. It is important to recognise that the objective throughout this thesis is not to evaluate whether tools can provide absolute truth about habitat condition, but rather to assess their ability to assess relative pressures on salmonid habitat from one location to the next. It is this ability to quantify relative risk which is most important to fisheries managers in enabling them to prioritise one location over another in respect of habitat restoration. A tendency in the past has been to assume that science will be able to predict exactly, all environmental variation. However, this may not be possible, practical or affordable. Instead, acknowledging uncertainty as an inherent feature of natural systems and incorporating it into management decisions is required (Milner *et al.*, 2003). In this respect considering habitat pressures in terms of their relative and probabilistic risk through space is the objective of this thesis.

4.2 Virtual walkover survey

The visual assessment of aerial imagery has been widely applied in the fields of cartography to produce digital maps, such as the Ordnance Survey's LandLine® and MasterMap® data sets (e.g. Cassettari, 2004); and in archaeology to survey and map historic remains (e.g. Bewley, 2003). This technique enables the precise location and quantification of feature presence and extent to be analysed, and is particularly suited to the mapping of linear features such as riparian habitat. For example, by mapping the extent of bank erosion, it will be possible to identify those tributaries within the Eden catchment that are under most pressure from intensive grazing. Within this chapter, aerial photography and GIS have been used to test the possibility for virtual geomorphological reconnaissance surveying, involving the mapping of riparian condition and channel features from visual interpretation of aerial imagery. The aim was to replicate the more traditional walkover methodology. The aerial mosaics described in Chapter Three have been analysed within ESRI ArcGIS V.9 using visual assessment undertaken by a single interpreter (myself) and mapped using vector-based GIS digitisation.

4.2.1 Methodology

The first stage of the assessment was to determine which variables should be surveyed. A review of existing walkover methodologies was first undertaken to establish the type of variables typically recorded and the various techniques and variable classification systems used to do so. This included evaluation of the Environment Agency's River Habitat Survey (RHS) (Environment Agency, 2003), the Scottish Fisheries Co-ordination Centre survey protocol (Puhr, 2001) and APEM's walkover survey of salmonid habitat (Hendry and Cragg-Hine (1997) in the context of the features and processes identified as ecologically relevant to salmonids within Chapter Two. In particular, variables were considered according to their potential for providing information on three key issues of concern to fisheries managers and habitat restoration practitioners such as the Eden Rivers Trust: (1) accelerated bank erosion due to agricultural stock access; (2) second the level of cover/shade provided by overhead riparian trees; and (3) the availability and distribution of suitable bed material for juvenile salmonids. Table 4.1 provides details of the variables selected and classification systems employed.

Table 4.1: *Features recorded during visual interpretation of aerial photography*

Feature	Code	Description	Code	Description
Bank erosion presence		Record the presence or absence of erosion separately for each bank. Assume erosion present if there is evidence of freshly exposed bank sediments without vegetation, failure or collapse, signs of livestock poaching or active un-vegetated point bars.		
	Y	There is evidence of erosion, either one large area or several smaller areas		
	N	There is no or only very slight evidence of erosion within the whole context of the management unit, e.g. slight erosion associated with a single tree could be ignored.		
	UK	The banks cannot be seen due to the presence of tree cover.		
Erosion Type1		Record the primary cause of erosion separately for each bank. If there is no erosion, leave blank. Due to the difficulties of viewing the bank profile from imagery a direct assessment of erosion causes should be made as described in the text		
	ST	Erosion is primarily due intensive grazing and agricultural stock access		
	FL	Erosion is primarily the result of natural fluvial processes such as meander migration.		
	TP	Erosion is primarily due to topographic failure.		
	TR	Erosion is primarily due to scour around tree roots.		
Erosion Type2		If there is a secondary contributory cause of erosion record it using the same codes as above.		
Certainty		Assess the level of certainty that the classification of erosion made is correct. A subjective measure newly introduced for the virtual walkover.		
	100	Very Certain		
	50	50% Certain		
	0	Very uncertain (For example, uncertainty may by greater for units with tree cover)		
Shading		Record the percentage of channel that has cover provided by riparian trees and vegetation (e.g. canopy and over hanging boughs) Based on RHS see text for further description		
	0	0-25% of channel cover	75	75-90% of channel cover
	25	25-50% of channel cover	90	90-100% of channel cover
	50	50-75% of channel cover		
Tree Density		Record the presence and density of tree cover separately for each bank. Based on a compressed version of the RHS classification		
	CO	Continuous or semi-continuous tree coverage		
	CL	Trees occur in clumps		
	SC	Trees are scattered along the bank or regularly but sparsely		
Stock		Record stock access. Assume access is present if the adjacent land cover is pasture and there is no visible evidence of a fence or riparian buffer to prevent access.		
	Y	Record yes if access from one or both banks.		
	N	Record no if no access from either bank		
Dominant land cover		Record the dominant land cover type within 50m of the bank. Based on RHS definitions		
	IP	Improved pasture	U	Urban/sub-urban (includes park/garden)
	RP	Rough pasture	MO	Moorland
	SC	Tall Herb and Scrub	B	Broadleaved woodland
	A	Arable (Tilled land)	C	Conifer plantation
Riparian land use		Record land use within the riparian zone (~5m from channel) using the same codes as above		
Substrate		Record channel bed substrate type if any evidence can be seen, e.g. from bar depositions, bedrock outcrops, or protruding boulders. Based on RHS definitions. See text for further details		
	BR	Bedrock	CGP	Cobbles, gravels and pebbles
	BB	Bedrock and boulders	Sa	Sand
	BO	Boulders	*(Si)	Any of the above followed by (Si) indicates siltation
Modified		Record any evidence of channel modification		
	S	Straightened		
	G	Groynes or Gabians		
	R	Bank reinforcement		
Deposition		Record any evidence of depositional bars Based on RHS definitions but for simplification no distinction is made between bar types (e.g. point bar, mid-channel bar)		
	UV	Unvegetated bars		
	VB	Vegetated bars		
	UK	Unknown (e.g. channel is obscured by riparian trees)		

4.2.1.1 Bank erosion assessment

Chapter Two identified the impacts of bank erosion upon salmonid populations as both positive and negative. The natural fluvial process of channel migration accompanied by bank erosion and deposition was considered to have mainly positive impacts for salmonids, although very rapid natural bank erosion could also have negative impacts. Accelerated bank erosion due to intensive grazing and stock access was considered to have negative impacts. Therefore, in addition to simply recording the presence of bank erosion, its presence on one or both banks within individual reaches was made as an indication of channel widening and accelerated erosion (both fluvial and stock related), reaches considered to exhibit particularly severe or excessive bank erosion were highlighted and an assessment of the causal process was made. Many walkover surveys record the type of bank erosion according to the type of failure (e.g. planar, slab, rotational) or bank profile (e.g. undercut, vertical earth cliff, composite) present. These features are then associated with particular erosive processes (Sear *et al.*, 2003). For example, composite banks are often interpreted as indicative of stock poaching. However, identification of such features was not considered feasible from the aerial imagery. Aerial photographs provide a vertical view of the channel, yielding information on features which occur in the horizontal plane such as planform geometry. Unfortunately, this makes it difficult to view features which occur in the vertical plane such as bank profile (Downs and Thorne, 1996). Instead a direct assessment of the dominant erosive process (e.g. stock, fluvial, topographic or scour around trees) has been made. This was aided by consideration of erosion location with respect to channel planform, the presence of depositional features, channel width, evidence of paleochannels and historic migratory activity within the floodplain, the adjacent land use within 50m and 5m, and stock access. All assessments were made in the context of both local geomorphic forms and in the wider context of the entire tributary in question, by using the multi-viewer capability offered within ArcGIS (Figure 4.1). Some examples of the type of information used to make assessments of erosion type are presented in Figures 4.2 and 4.3. It is possible that more than one erosive process may be operating within a reach. To accommodate this, up to two different processes could be recorded. However, there was still uncertainty attached to the assessment of erosion. Therefore, to improve classification knowledge, a subjective measure of certainty that erosion presence had been correctly identified was also recorded for each site, where 0=very uncertain, 50=uncertain and 100= certain.

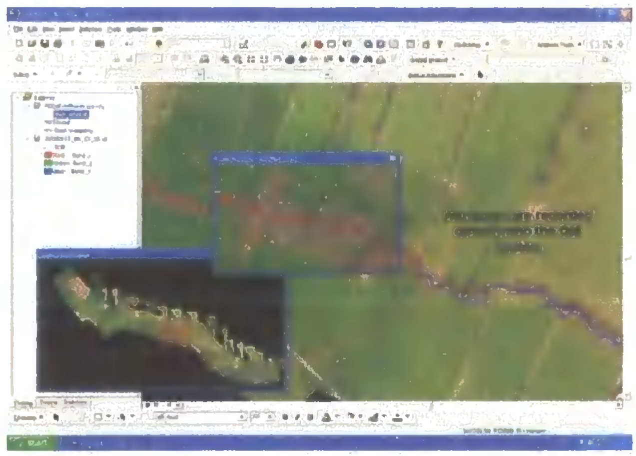


Figure 4.1: *Multi-viewer capability within ArcGIS enables river segments to be analysed at a range of scales simultaneously.*

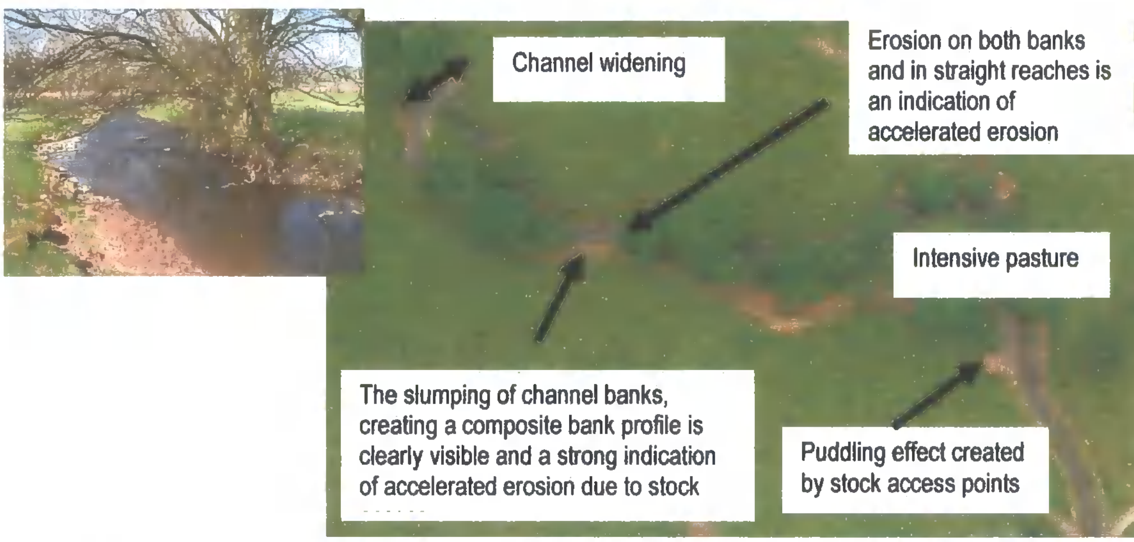


Figure 4.2: *Demonstration of bank erosion assessment using aerial photographs where the primary cause of erosion is intensive grazing*

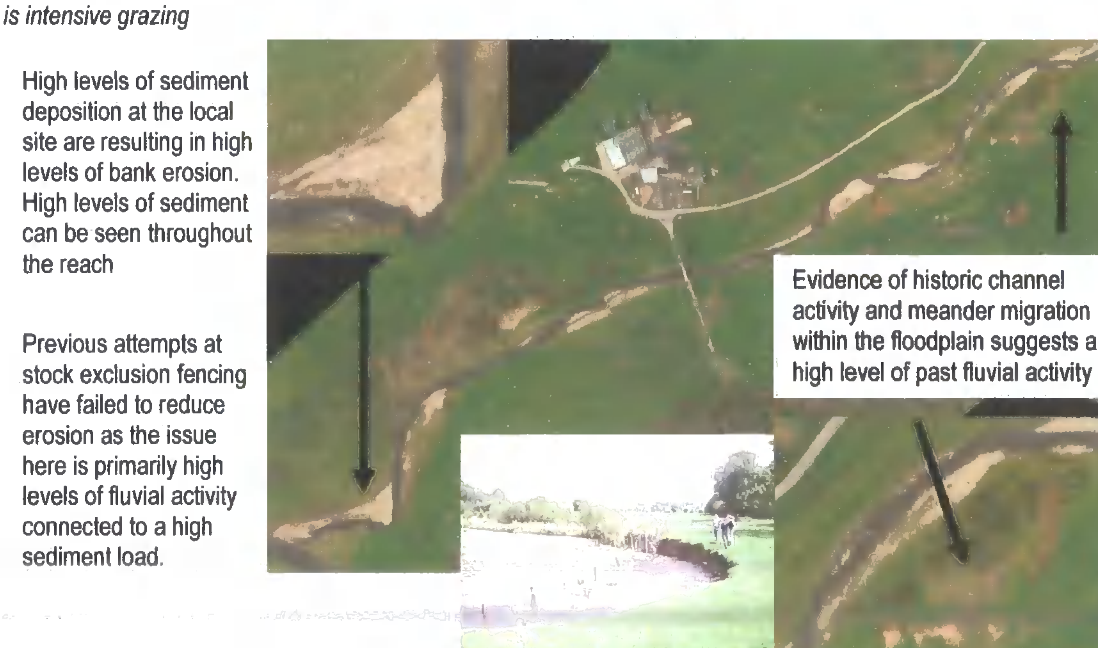


Figure 4.3: *Demonstration of bank erosion assessment using aerial photographs where the primary cause is related to fluvial processes and sediment regime.*

4.2.1.2 Shade/cover assessment

As with bank erosion, Chapter Two noted both positive and negative impacts of riparian trees upon salmonid habitat. However, there is little information within the scientific literature as to the optimum level of overhead cover required by salmon and trout, or the level at which impacts become negative or positive (Armstrong *et al.*, 2003). To provide more information on this factor, the percentage of the channel in each reach covered by overhead vegetation was recorded (the shading variable). This included cover provided by canopy vegetation and over hanging boughs, both of which are typically recorded during an RHS survey. However, rather than simply recording presence or greater than 33% cover as in RHS, an attempt to more precisely estimate the average level of cover (<25%, 25-50%, 50-75%, 75-90%, >90%) was made (Figure 4.4). The proportion of channel covered was estimated throughout the entire reach regardless of tree distribution. The amount of additional cover likely to be provided by leaves was taken into account when estimating cover for images captured prior to foliage development. No attempt was made to estimate the amount of in-stream cover provided by submerged vegetation, tree roots, boulders or woody debris as this was not visible within the imagery. A second variable was then recorded to classify the dominant form of tree distribution (tree density) observed on each bank. This was based on the definitions of tree distribution included in the RHS manual, although the classes were amalgamated for simplification as presented in Table 4.1. In addition, information regarding tree type (coniferous or broadleaved) could be determined from the riparian land use category (Table 4.1). This is important as tree type will affect the type of organic matter and terrestrial invertebrates delivered to the channel, and subsequently the impact of tree cover on salmonid populations. For example, deciduous woodland is considered to support a greater diversity and abundance of invertebrates than coniferous woodland (Allan *et al.*, 2003).

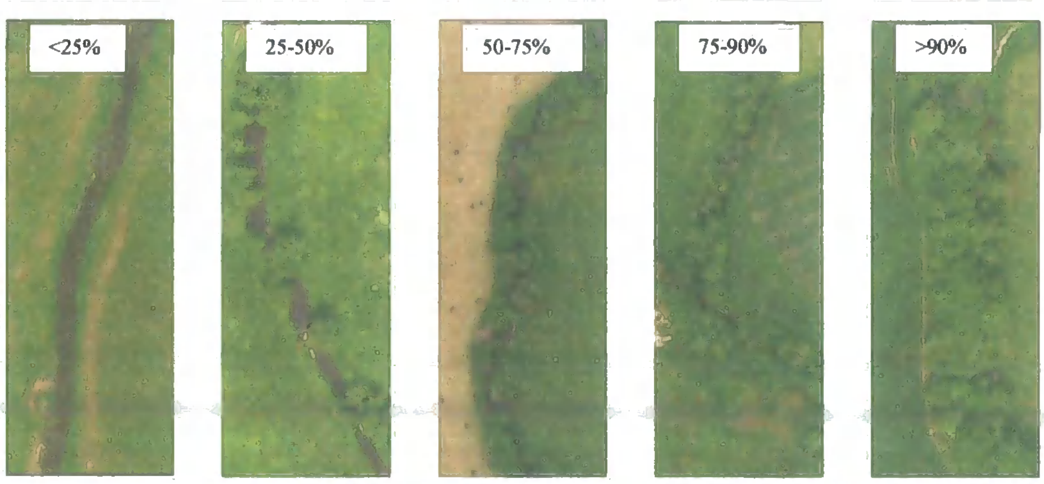


Figure 4.4: *Example images illustrating the classification of channel cover*

4.2.1.3 Physical in-stream habitat assessment

Traditional walkover surveys typically classify channel substrate according to size. A variety of different methodologies are applied within the literature. Some surveys will sample a set number of clasts from within the reach and measure their dimensions to give an objective estimate of the median size (D_{50}) (Sear et al., 2003). Alternatively, others may use a more subjective estimate, of either the dominant substrate size or proportion of the bed covered by each of the substrate sizes (Environment Agency, 2003). Here the broad channel substrate class, as identified from bar deposits, bedrock outcrops or protruding boulders was recorded in accordance with the definitions provided in the RHS manual. The aim of this classification was to provide indicative information on those areas of the catchment that are most suitable for supporting salmonid spawning and juvenile salmonid populations due to the presence of gravel, pebble and cobble substrate. Unfortunately, it was not possible to precisely estimate substrate size visually and in particular, it was not possible to distinguish between gravel, pebble and cobble categories. As such, visual distinction between spawning, fry and parr habitat was also not possible. However, it was possible in areas of exposed substrate to distinguish gravels, pebbles and cobbles from bedrock, boulder or sandy/silty substrates which are considered less suitable for salmonids of all life stages (Figure 4.5).

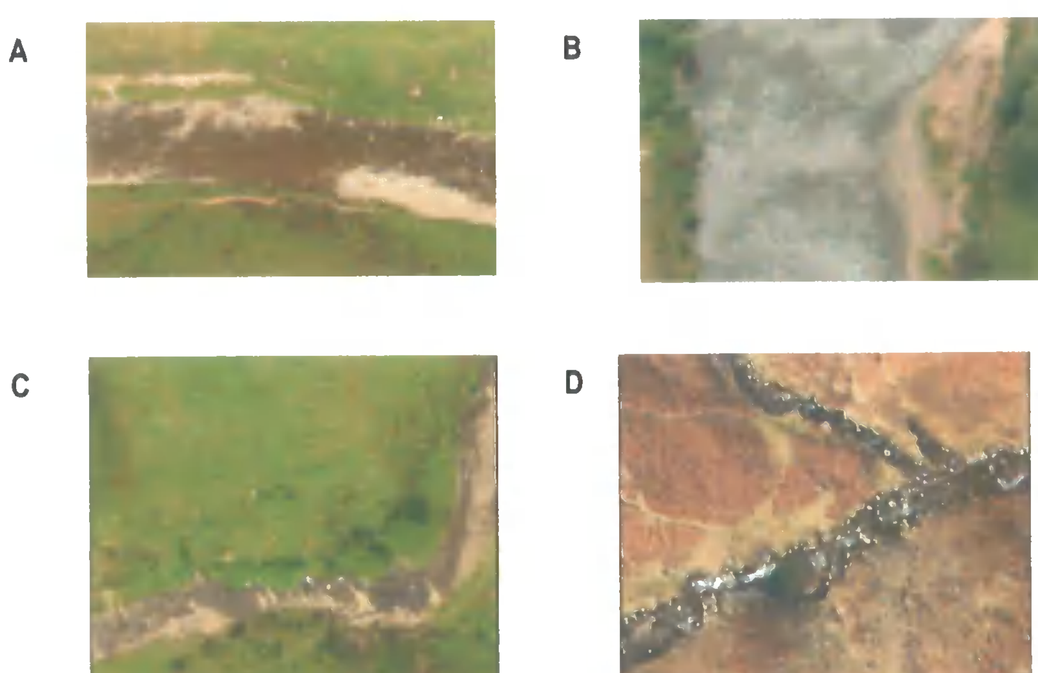


Figure 4.5: *Example images illustrating the classification of broad substrate type (a) cobble/gravel/pebble (b) sand (c) bedrock (d) boulder*

Channel substrate has been classified in more detail within large river channels using automated image processing procedures and very high resolution (1-3cm) aerial imagery (Carbonneau *et al.*, 2004). However, the 20cm resolution data used within this research, and the fact that many of the streams within the Eden catchment are relatively narrow (< 10m wide) and shaded, meant that applying similar techniques was not considered feasible here. Hydraulic conditions and flow type are also frequently recorded during walkover surveys, and coupled with substrate to identify areas of potential salmonid habitat. However, following assessment of the raw imagery, it was concluded that these features could not be readily mapped visually. The ability to identify flow type using automated classification processes and DTM processing is evaluated later within this chapter.

4.2.1.4 Feature mapping

Following variable selection, the next stage of the assessment was to determine how best to record the variables within GIS. Walkover surveys record features by a number of methods including: (1) physical mapping of the precise location and extent of each feature on to paper maps in the field and transferring the information to GIS using a combination of vector polygons, polylines and points (e.g. Hendry *et al.*, 1997); (2) by sampling information at individual sites of known geographic location recorded using GPS equipment (e.g. Parsons *et al.*, 2001) and representing information in a GIS using vector points; or (3) by recording information for individual but continuous lengths of river with known start and end locations and representing information in GIS using segmented vector lines. Rivers are linear, continuous features, and knowledge of the precise extent and location of environmental pressures is important for managers; therefore, either approaches (1) or (3) are considered most applicable.

It was decided to use option (3) as this method represents a balance between continuous recording of features and speed. Details of the selected variables ('attributes' in GIS terminology) have therefore been attached to individual sections ('arcs') of a segmented river line using the codes presented in Table 4.1. A number of different river lines were available in digital format, including OS Land-Line® data and the CHASM river centreline digitised from OS 1:50,000 raster data. However, on examination of these datasets, river centrelines were found to be displaced in a number of locations from the position of the channel as shown in the aerial imagery (Figure 4.6). This may be because: (1) the river channel has migrated since the OS data were mapped; (2) the digitisation of the CHASM centre-line at a scale of 1:50,000 is too coarse; or (3) there is error in the geo-referencing of the aerial imagery. In addition, OS Land-Line® data records a double 'blue line' (digitised for each bank) where channels are greater than 2m wide rather than a single centreline. To overcome these issues a new river centreline was digitised in alignment with the aerial imagery.

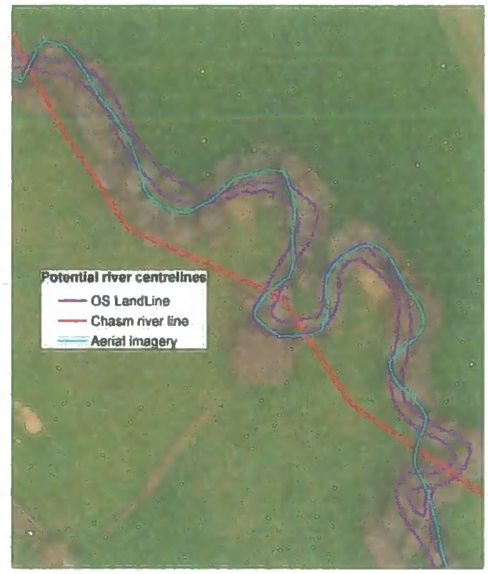


Figure 4.6: *Variations in river centreline position compared with aerial photographs*

It was decided to segment the river line according to the scale of land management present, typically the field scale. This is the scale at which habitat restoration is generally undertaken and it is therefore information at this scale which is most relevant to managers. Where land management units did not correspond between the left and right bank, the river line was segmented according to the smallest unit present. Units were aggregated for the purposes of assessment providing they were all under the same type of land management and exhibited the same characteristics. All mosaics captured during the 2004 aerial survey were analysed. 2769 land management units in total were classified for the River Eden and its tributaries, totalling 658km in length, a process which took one interpreter approximately 50 working days to complete. Attaching attributes to river segments in this manner has enabled data queries to be performed highlighting and calculating the length of river reaches subject to a particular factor or combination of factors. This is an important capability for practitioners who need to estimate the amount and cost of restoration projects required (e.g. the amount of riparian stock exclusion fencing required).

4.2.2 Validation of virtual walkover survey data

To enable validation and accuracy assessment of the image analysis, ground truth data were sampled at 104 sites across the catchment. 54 of the sites were sampled at the same time and location as the 2005 quantitative electrofishing survey (Section 3.3.2). These sites were specifically selected by the Eden Rivers Trust's Fisheries Officer. Site selection was made using the analysed GIS data to select sites the represented a variety of habitat types across the full range of variables surveyed. The remaining 50 sites were randomly selected using the random number function available in Microsoft Excel and the unique ID code applied to each management unit during GIS analysis. The entire length of each selected validation unit was walked before the assessment made and exactly the same features were recorded, in the same manner, as during image analysis. The validation units surveyed corresponded exactly in length to the management units analysed using GIS. Variables for were recorded onto a survey sheet making a note of the corresponding management unit ID code. This enabled site matching and accuracy assessments to be made. All ground truth sites surveyed were included in the accuracy assessment. Geomorphological reconnaissance surveys are typically subject to the subjective decision making of individual surveyors (Downs and Thorne, 1996), and this survey is no exception. To try to reduce subjective error, all virtual assessments have been made by a single interpreter (me) who is familiar with the use of aerial photography and who has previous fluvial geomorphological knowledge. However, ground truth data were collected both by the interpreter (me) who analysed the imagery and another surveyor who I had trained. Using a different surveyor to collect a considerable amount of the validation data reduced the introduction of bias due to the interpreter having previously examined aerial imagery of the site. It also allows assessment of the technique's robustness to observer subjectivity, although this may have caused operator differences within the validation data. All validation data were collected during summer 2005 following assessment of the aerial imagery. As the variables recorded are considered representative of the range of features typically recorded by walkover surveys it has been possible to make direct comparisons between the two techniques. In particular, comparisons have been made between the results of this survey and the River Eden, River Habitat Survey undertaken for part of the upper catchment by the Environment Agency (Parsons *et al.*, 2001).

4.2.3 Accuracy assessment

Validation of the virtual walkover methodology and the other two methodologies presented within this chapter has been undertaken using well established, formal accuracy assessment procedures (after Congalton & Green, 1999), based upon the error matrix (Figure 4.7). These techniques assess the accuracy of classifications made using remotely sensed data by comparing them to either ground truth data or, in the case of automatically classified imagery, to visual assessment of the imagery, at the same geographic location.

j=columns (validation data)

i=rows (classified data)

	1	2	k	n_{i+}
1	n_{11}	n_{12}	n_{1k}	n_{1+}
2	n_{21}	n_{22}	n_{2k}	n_{2+}
k	n_{k1}	n_{k2}	n_{kk}	n_{k+}
n_{+j}	n_{+1}	n_{+2}	n_{+k}	n

Figure 4.7: *The error matrix (After Congalton and Green, 1999)*

Three principal accuracy statistics have been computed. First, the overall accuracy [4.1] was calculated as:

$$\text{Overall accuracy} = \frac{\sum_{i=1}^{k} n_{ii}}{n} \qquad [4.1]$$

by dividing the number of correct classifications made (Σn_{ii}) by the total number of classifications made (n) for each category. This is the most commonly presented accuracy statistic. However, this assessment may over-estimate the level of agreement achieved as it does not take into account the expected levels of chance agreement given the sub sample sizes in use. To account for this a second measure of accuracy the Kappa (or Kappa-Hat) statistic was also computed [4.2] which is often cited as a more reliable measure of accuracy or agreement than overall accuracy (Cohen 1960):

$$\hat{K} = \frac{n\sum_{i=1}^{k} n_{ii} - \sum_{i=1}^{k} n_{i+}n_{+i}}{n^2 - \sum_{i=1}^{k} n_{i+}n_{+i}} \qquad [4.2]$$

where $$n_{i+} = \sum_{j=1}^{k} n_{ij} \quad \text{and} \quad n_{+i} = \sum_{i=1}^{k} n_{ij} \qquad [4.3 \text{ and } 4.4]$$

[4.2] is essentially expressing the ratio of the observed excess over chance agreement to the maximum possible excess over chance, with Kappa = 1.0 at perfect agreement and Kappa = 0.0 when observed agreement equals chance agreement (Everitt, 1998). To aid interpretation of the

Kappa statistic Table 4.2 presents a subjective classification of Kappa values as recommended by Altman, (1981).

Table 4.2: *Values of Kappa and their strength of agreement (After Altman, 1981)*

Value of Kappa	Strength of agreement
<0.20	Poor
0.21-0.40	Fair
0.41-0.60	Moderate
0.61-0.80	Good
0.81-1.00	Very good

Both the overall accuracy and Kappa statistic report on the integrated accuracy across all classes within an error matrix. However, the ability to classify one particular category may be more important than another. For example, using the virtual walkover survey methodology, the ability to identify erosion due to intensive grazing and stock access was considered more important than the ability to identify erosion due to topographic failure processes. In order to assess the accuracy of individual classes, a third measure of accuracy, the Conditional Kappa statistic, was used [4.5]:

$$\hat{K}_i = \frac{n_{ii} - \left(\frac{n_{i+}n_{+i}}{n}\right)}{n_{i+} - \left(\frac{n_{i+}n_{+i}}{n}\right)} = \frac{nn_{ii} - n_{i+}n_{+i}}{nn_{i+} - n_{i+}n_{+i}} \qquad [4.5]$$

This computes the maximum likelihood estimate of the Kappa coefficient for the conditional agreement for the *i*th category. As for the Kappa statistic the Conditional Kappa also corrects for chance agreement and can also be interpreted using Altman's classification system.

4.2.4 Accuracy assessment for the virtual walkover assessment

Table 4.3 presents the results of the formal accuracy assessment and shows that results were extremely promising, particularly for the two categories most likely to be of interest to fisheries managers, bank erosion and tree cover, with all their associated categories reporting moderate or good agreement strengths. Unsurprisingly, the severe erosion category reports the best agreement as by its very nature it creates the most observable mark upon the imagery. However, it should be noted that as the feature category scale decreased, the ability to observe and to classify it remotely became more difficult and a decrease in the Kappa value was observed. The ability to classify features within these smaller scale categories was typically diminished by one or more of the following three factors. First, the images were near-vertical and observations of features in the vertical plane such as the bank face and bank protection measures were more problematic. Second, the presence of tree cover had its greatest impact upon small in-stream features, for example, it could totally obscure the presence of depositional bars. Third, in-stream features such as substrate also became obscured as water depth and water turbidity increased. However, the Kappa value (0.448) for substrate (without siltation), probably the next feature (after erosion and tree cover) of most interest to fisheries managers did still achieve a moderate agreement, although decreasing feature scale within this category to include siltation did result in a decrease in accuracy to a Kappa value of 0.385.

Table 4.3: *Accuracy assessment statistics for the virtual walkover methodology*

Feature	Overall Accuracy (%)	Kappa	Strength of agreement
Erosion presence	78.4	0.569	Moderate
Primary erosion type	70.7	0.571	Moderate
Severe erosion	89.4	0.641	Good
Stock access	76.9	0.481	Moderate
% of channel cover	66.3	0.481	Moderate
Tree density	63.9	0.487	Moderate
Land cover (50m)	Not enough validation sites for analysis		
Riparian land use (5m)	63.9	0.282	Fair
Channel modification	84.6	0.217	Fair
Substrate (with silt)	60.3	0.385	Fair
Substrate (without silt)	69.8	0.448	Moderate
Depositional features	55.8	0.200	Poor

Concerns over the use of aerial photography for providing morphological information about river channels have been raised within the scientific literature, particularly, with reference to the presence of woody riparian vegetation which may obscure channel banks, especially in small channels (Downs and Thorne, 1996). As noted in Chapter Three, the majority of the imagery used was collected prior to development of full leaf foliage. However, there were still a number of land management units (approximately 20%) where bank erosion presence on one or both banks was classified as 'unknown' due to a restricted view caused by the presence of trees. For the purposes of the accuracy assessment above, it was assumed that there was no bank erosion in the aerial photography measurements of these units. Results suggest that the level of error in the analysis was not degraded by this assumption. Therefore, the presence of riparian trees was not considered to limit the use of aerial photography in assessing bank erosion presence within the Eden catchment, although identification of smaller scale features, such as point bars, may be affected. A second concern was that the imagery would no longer be applicable following a large flood event. Such an event occurred in the Eden catchment during January 2005 between the dates of image capture and validation. The results showed that classifications remained robust to extreme flow events.

Classifying bank erosion was one of the main priorities of this methodology and a measure of certainty that erosion presence had been correctly identified was also recorded to improve classification knowledge, where 0=very uncertain, 50=uncertain and 100=certain. Table 4.4 shows that including this parameter improved classification accuracy from the total dataset to the certain category from moderate to good. It should also be noted, as indicated by the Conditional Kappa statistics, that it was the erosion present category which was most sensitive to (i.e. most improved by) this stratification. As this was the feature of greatest interest, inclusion of the certainty parameter in future classifications is highly recommended. Including a measure of certainty such as this enables managers to stratify erosion classification maps into areas where they are certain there is erosion and areas where a ground survey may be necessary to first confirm the presence and extent of erosion before management action can be targeted. This allows managers to allocate resources such as a walkover survey team more efficiently and cost effectively.

Table 4.4: Accuracy assessment for erosion presence stratified by classification certainty

Feature	Overall Accuracy (%)	Kappa	Strength of agreement	Conditional Kappa	
				Absent	Present
Total dataset	78.4	0.569	Moderate	0.695	0.481
Certain	81.6	0.630	Good	0.697	0.575
Uncertain	70.5	0.432	Moderate	0.695	0.313

The ability to identify the cause of erosion was also considered important, as this has been acknowledged to determine the impact of bank erosion (positive or negative) upon salmonid populations. In particular, identifying reaches where bank erosion had been accelerated by intensive grazing and agricultural stock access was important due to its widely perceived negative impact upon in-stream ecology in general and salmonids in particular. Table 4.5 presents the accuracy statistics calculated for the individual erosion types under the following 4 scenarios:

(1) predicted primary erosion type is observed as the primary erosion type;

(2) predicted primary erosion type is observed as either the primary or secondary erosion type;

(3) predicted primary erosion type is observed as primary erosion type but only analysing those categories where erosion presence was correctly identified; and

(4) predicted primary erosion type is observed as either the primary or secondary erosion type but only analysing those categories where erosion presence was correctly identified.

Table 4.5: *Accuracy assessment for erosion type*

Feature	Overall Accuracy (%)	Kappa	Strength of agreement	Conditional Kappa			
				Fluvial	Stock	Topographic	Tree scour
1	70.7	0.571	Moderate	0.433	0.484	0.588	0.223
2	75.7	0.638	Good	0.507	0.660	0.592	0.243
3	78.4	0.643	Good	0.675	0.615	0.564	1.000
4	91.9	0.859	Very good	0.842	0.944	0.577	1.000

Fluvial, stock and topographic erosion types were all classified relatively well under all 4 scenarios with Conditional Kappa values ranging from 0.433 to 0.944. Only the identification of tree scour was poorly classified. This is unsurprising due to the restricted view created by trees in these reaches. If identification of tree scour is important for managers, they could alternatively use the aerial imagery to target tributaries with a high proportion of tree-lined banks for walkover surveys. As would be expected, accuracy generally increased from scenario 1 to 4; so that where erosion presence was correctly classified, the type of erosion was also correctly identified with a high level of accuracy. For management purposes it is probably not necessary for the primary erosion type to be classified exactly as the primary erosion type (scenarios 1 & 3). Instead, recognition of contributing factors in any order will probably suffice (scenarios 2 & 4). In terms of recognising agricultural stock access and riparian grazing as a contributing factor to bank erosion, Conditional Kappa statistics were very promising ranging from 0.660 (good agreement) for all reaches to 0.944 (very good agreement) for those reaches where erosion presence was correctly identified.

The ability to classify the degree of overhead cover/shade provided by riparian trees and in particular, the extreme high and low categories, was also considered a major priority of this research. Table 4.6 shows that the, <25%, 75-90% and >90% shade categories are predicted well with Conditional Kappas of 0.801, 0.649 and 0.606 respectively. However, a lower level of accuracy was reported for the 25-50% and 50-75% categories. This is unsurprising as classification error can occur in two directions compared with only one direction for the most extreme categories. Additionally, incorrect classifications were typically only in error by an order of one class. If these classes are merged the Kappa statistic (0.975) records a near perfect agreement between the classified and observed data, indicating that aerial photography can be used very reliably to identify the amount of overhead cover provided by trees.

Table 4.6: Accuracy assessment for channel cover/shade

Feature	Overall accuracy	Kappa	Strength of agreement	Conditional Kappa				
				0-25	*25-50*	*50-75*	*75-90*	*90-100*
1	66.3	0.481	Moderate	0.801	0.367	0.278	0.649	0.606
2	98.1	0.975	Very good	1.000	0.879	1.000	1.000	1.000

1: Accuracy to the exact class

2: Accuracy to the nearest two classes

4.2.5 Comparison of virtual and traditional walkover methodologies

A principal driver for developing the virtual walkover methodology was that the traditional tools available e.g. the ground reconnaissance survey, were considered prohibitively time consuming and costly for assessing river habitat at the catchment-scale. It was therefore important to evaluate the virtual technique from a practical perspective in relation to more traditional ground surveys and a number of major advantages are considered to apply. First, the virtual walkover methodology facilitates rapid coverage of an extensive area. Analysis of 658km of river and riparian habitat took one single interpreter 50 days to complete. A walkover survey would take considerably longer and most organisations would insist on this being done by two people for health and safety reasons (Table 4.7).

Table 4.7: *Comparison of costs for different reconnaissance surveys*

Cost	Virtual reconnaissance survey	River Habitat Survey† (1 surveyor used)	River Habitat Survey (2 surveyors used)	APEM's rapid fisheries survey‡
Time (man days)	50 days	188-263 days plus data entry time	376-556 days plus data entry time	165 days plus data entry time
Monetary cost	~£50k for photography plus analyst time	£80-180k		Unknown

† Based on information from the Environment Agency, the one 500m site can be surveyed in 1 hour and costs £40-£120 plus £20 for data entry; my own assumption is that five to seven sites could therefore be surveyed in a day.
‡Based on information from APEM Ltd that 4km can be surveyed per day by one surveyor.

Second, reach attributes were recorded directly into the GIS enabling immediate spatial analysis rather than the need for data entry when surveyors returned from the field. Third, it enabled data collection in areas otherwise inaccessible due to access restraints or difficult terrain, thereby providing a continuous data source. Fourth, individual management units can be evaluated in terms of the geomorphological context of an entire reach or tributary by zooming in and out of the imagery or by using multi-viewer capability at a range of scales. Walkover surveys are subject to bias by what is within the surveyor's field of vision at the time they complete the survey form.

Fifth, reaches sometimes many kilometres apart can be compared next to each other on the screen thereby reducing errors due to subjectivity. Sixth, imagery provides a powerful tool for explaining environmental issues to the public, helping to encourage co-operation from local communities, landowners and funders in restoration projects; and seventh, it provides a permanent visual record of the riparian corridor that can be revisited to extract additional information, gather further expert opinion, and train/improve surveyor's capability.

The results of the virtual walkover survey have been compared with those of the Environment Agency's River Habitat Survey (RHS) undertaken for 5 tributaries (River Belah, River Eamont, Hilton Beck, River Lowther and Scandal Beck) of the River Eden (Parsons *et al.*, 2001). Whilst it is difficult to make exact comparisons due to the different nature of the surveys, some of the general conclusions have been compared. First, RHS reported that overall, erosion due to stock poaching was present at 33% of sites surveyed. A comparatively similar result was found with the virtual walkover, which reported 29% of the channel surveyed to have banks eroding due to stock access. Second, in terms of cover levels, RHS reported that channel shading was present at approximately half of the surveyed sites. Results from the virtual survey were similar with 60% of the area having some shade, 28% having >50% shade and 5% having greater than 90% shade. Similar results were also found in terms of substrate dominance (Table 4.8).

Table 4.8: *Comparison of dominant substrate classifications generated from EA RHS (Parsons et al., 2001) and the virtual walkover technique.*

River	RHS	Virtual walkover
Belah	Cobble	Cobble, gravel, pebble
Eamont	Cobble	Cobble, gravel, pebble
Hilton	Boulder	Cobble, gravel, pebble
Lowther	Boulder and cobble	Boulder, cobble
Scandal	Cobble	Cobble, gravel, pebble

Whilst many of the general characterisations on a tributary basis are similar, the advantage of the virtual walkover survey in comparison with RHS is that features are recorded continuously, rather than being sampled every 500m. The RHS sampling protocol may capture the general character of a particular tributary but the results of the virtual walkover technique are considered more useful to fisheries managers as they allow the precise extent and location of pressures to be identified and the costs of restoration required to be calculated.

However, virtual surveys cannot provide all the detail of a ground reconnaissance survey. As results showed, accuracy was lower for fine-scale features such as depositional bars and bank protection. For surveys where this information is essential, the walkover technique may still be preferable. Additionally, whilst the cause of erosion and therefore management action was readily identifiable in the case of agricultural grazing and stock damage, where the source of the problem was located adjacent to the effect, identifying the cause and therefore any necessary remedial action, is often more problematic in terms of fluvial erosion. This is because the source such as increased flows or a change in sediment regime upstream may often be distributed in extent and located many kilometres from the resulting effect. In these cases, sustainable solutions will only be identified through the use of walkover surveys undertaken by trained geomorphologists. However, what the remote sensing survey can do is highlight reaches where there is considerable fluvial erosion that may be problematic, for example where both banks are eroding, indicating a river that is widening, enabling managers to target walkover team resources more cost effectively.

4.3 Automated classification of relative water depths

Another variable frequently included in walkover surveys, particularly those designed for ecological purposes, is in-stream habitat type. As described in Chapter Two, in-stream habitat typically comprises velocity, depth and substrate. These raw variables may be recorded during detailed, small scale, ground surveys, but due to the time and equipment costs associated with measuring these variables, field survey techniques are not applicable at the catchment-scale. Alternatively, most field reconnaissance surveys employ surrogate variables, which can be mapped visually to categorise in-stream habitat. In user-specific surveys variables based on the species of interest e.g. salmonid spawning habitat, juvenile (fry/parr) habitat (Hendry and Cragg-Hine, 1997) may be used. Generic surveys typically record variables such as flow type e.g. broken standing wave (Environment Agency, 2003) or for more ecological meaning physical biotope e.g. riffle/pool (Padmore, 1998).

The potential for collecting both raw and surrogate habitat variables using remotely sense data and automated image classification procedures has been demonstrated by a number of researchers. Puestow *et al.* (2001) used 1.5m CASI multi-spectral imagery to distinguish between areas of riffle/rapid and areas of run/steady/flat in the Come by Chance River, Newfoundland, Canada, reporting an overall accuracy of 64.4%. Higher resolution, 1m hyper-spectral (128 band) imagery was employed by Marcus *et al.* (2003) to map in-stream habitat (glides, riffles, pools and eddy drop zones) with an overall accuracy ranging from 69% for third order streams to 86% for fifth order streams in the Lamar River, USA. With regard to the estimation of raw habitat variables, one of the most widely applied algorithms for relating reflectance values to water depth was developed by Lyzenga (1981), using 8m multi-spectral data to estimate depths in the shallow coastal waters of the Great Bahama Bank. Within a river environment, Gilvear *et al.* (1995) used 1m scanned panchromatic photos to classify relative depth. The calculation of actual depths within habitat types has been demonstrated by Marcus *et al.* (2003) using 1m hyper-spectral imagery, who reported R^2 values ranging 28% for runs in third order streams to 99% for high gradient riffles in fifth order streams. Carbonneau *et al.* (2004) also mapped water depth and substrate for an 80km stretch of the Sainte-Marguerite River in Quebec, Canada, but used high spatial (1-3cm) colour photography as opposed to high spectral imagery reporting R^2 values of 40-60% on an individual image basis.

The previous research described has generally been undertaken for relatively short reaches (10-100km) in relatively wide, high order rivers, using either very high spatial or spectral imagery. As yet, the large scale application of such techniques has not been possible due to cost and practicality. However, McGinnity *et al.* (2002) have promoted the use of relatively cost effective 50cm resolution aerial photography for the mapping of in-stream habitat, and demonstrated the technique for a 3km reach of the Burrishoole River, Ireland. They used unsupervised classification, and labelled the resulting classes as various in-stream habitat types based on expert consultation and visual observation of both the classified and original imagery. The aim here is to evaluate this technique further to assess whether 20cm aerial photography and automated image processing techniques can be used to extract information regarding in-stream habitat, in particular relative water depth for small, low order streams within the Eden catchment.

4.3.1 Image classification methodology

The theoretical basis for extracting depth and in-stream habitat information from aerial imagery is that different habitat types exhibit different reflectance properties based on their water depth and turbulence, which result in different spectral signatures within the imagery. In terms of water depth, light is absorbed as it passes through the water column. The deeper the water the greater the level of absorption and hence the lower the amount of light reflected from the channel bed. In areas of shallow water, less light is absorbed and hence more light is reflected. This is represented in the imagery as differences in the brightness level or digital number (DN) of each pixel for each wavelength band (Red, Green and Blue). Shallow water therefore results in a higher DN and brighter colour than deep water. The relationship between water depth and reflectance can be expressed as:

$$I_{out} = I_{in}\, e^{-cx} \qquad [4.6]$$

where: I_{in} is the intensity of incoming light (or brightness level of the channel bed)

I_{out} is the intensity of outgoing light (or observed brightness level in the image)

x is the water depth

c is the rate of light absorption

(Carbonneau *et al.*, 2004)

The determination of water depth therefore relies on the ability to calibrate accurately, the rate of absorption and the initial brightness level, which will vary according to properties such as water turbidity and geology. Unfortunately, calibration of this relationship relies on the simultaneous collection of aerial imagery and a considerable quantity of ground validation data on water depth at known locations. Different calibrations are required for reaches of different substrates and turbidity. The collection of such data across a catchment as large and diverse as the Eden was simply not feasible. This is widely recognised as a major limitation in applying remote sensing technology to the estimation of water depth across large areas (Legleiter *et al.*, 2004). Instead, the ability to classify relative depth variations according to variation in spectral signature has been applied here. Initial investigations were undertaken using individual raw, composite RGB images or segments of images. A sample of four images from across the catchment, representing different stream types, sizes, location and time of image capture were selected and the following classification procedure, after McGinnity *et al.* (2002) was applied:

(1): Images were first clipped in ArcGIS so that only the river channel remained (Figure 4.8a). This is a commonly adopted procedure which enables the classification process to focus solely on the features of interest and avoid complications by excluding the surrounding landscape (e.g. Puestow *et al.*, 2001; Leglieter *et al.*, 2002; McGinnity *et al.*, 2002).

(2): Images were then imported to Erdas Imagine 8.1 for image classification. An unsupervised classification was applied on a per pixel basis, using the software's ISODATA algorithm. This defines distinct image classes based upon the stream's inherent spectral variability, rather than depending on the accuracy of field or visually determined training datasets as required by supervised classification (Leglieter *et al.*, 2002). It is also much faster and therefore more applicable at a catchment-scale than supervised classification. Several different classifications were run, each with a different number of classes. Based on visual assessment of the resulting thematic images, the classification scheme that represented the best compromise between

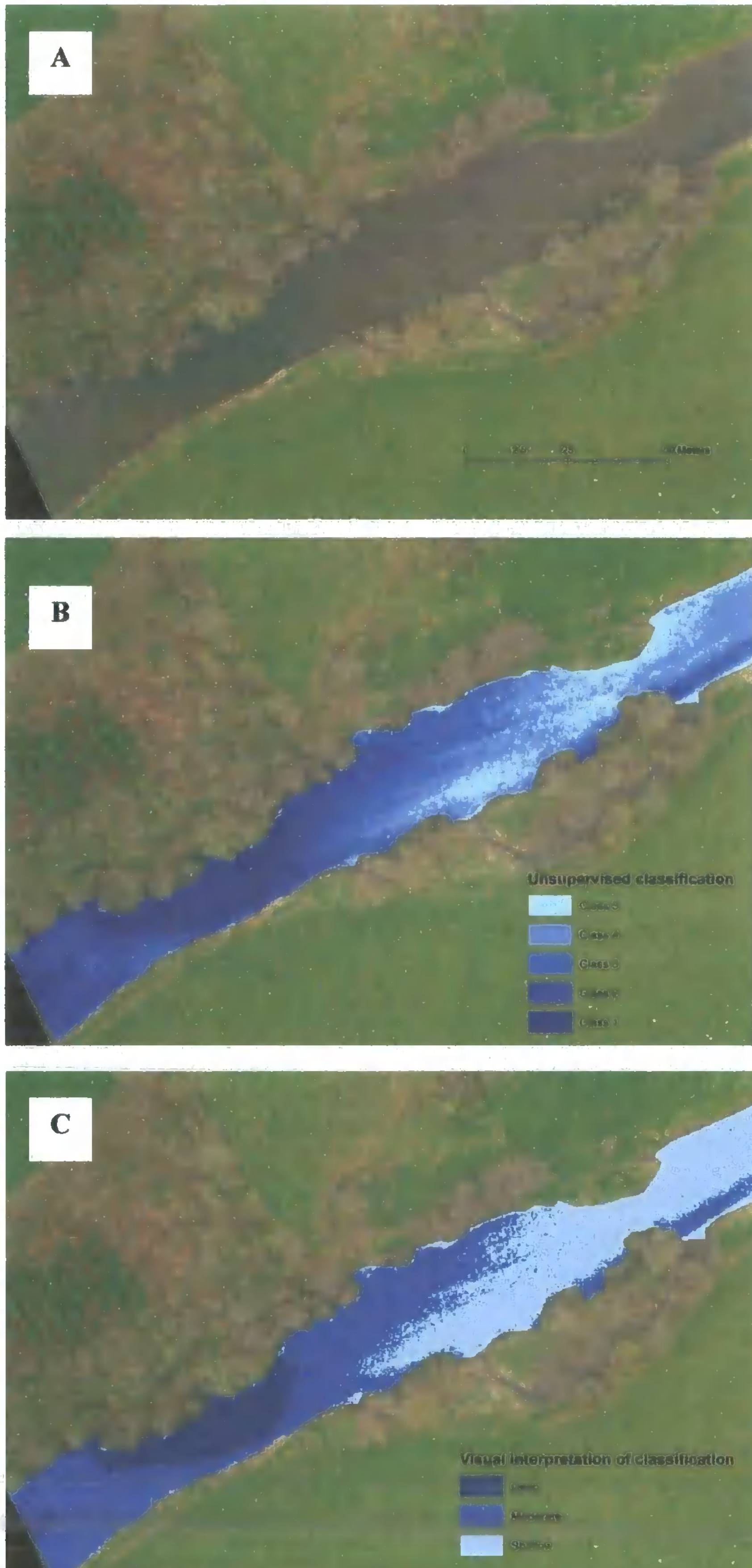


Figure 4.8: *Classification of aerial photographs using Erdas Imagine and unsupervised classification*

simplification and required separation of features was selected for each image. This typically had between 5 to 7 classes (Figure 4.8b).

(3): The classified image was then exported back into ArcGIS and simplified into the following four classes, deep, moderate, or shallow water, and where applicable exposed substrate, using visual assessment and the 'Reclass' function in Spatial Analyst. Classification was kept relatively simplistic rather than attempting a more detailed interpretation that would likely lead to misleading accuracy and precision in results (Figure 4.8c).

4.3.2 Validation of relative water depth analysis

Validation of the water depth classifications was undertaken using standard error matrix procedures as above (Table 4.9). A number of researchers have also suggested that field data may be less accurate than the classified data, as field surveyors lump spatially variable habitat types into large homogenous polygons that miss the fine scale heterogeneity captured by individual pixel classifications (Legleiter *et al.*, 2002), Instead, validation data were primarily generated by classifying 100 randomly selected pixels in every image by eye.

Table 4.9: *Summary of classification accuracies for relative water depths*

River	Overall Accuracy (%)	Kappa	Conditional Kappa			
			Deep	Medium	Shallow	Substrate
River Caldew	69	0.58	0.68	0.44	0.43	1
Helm Beck	79	0.72	0.80	0.50	0.78	1
Trout Beck	63	0.37	0.17	0.30	0.82	0.38
River Eamont	78	0.65	0.84	0.49	0.73	N/A

Overall accuracies, Kappa, and Conditional Kappa statistics for each of the four images were extremely promising, indicating the potential for using 20cm aerial photography and automated image processing procedures for the classification of relative water depth in low order streams. Overall accuracies of 63-79% and Kappa values of 0.37 (Fair agreement strength) to 0.72 (Good agreement strength) were recorded. These accuracies compare favourably with those reported previously using hyper and multi spectral data on large rivers (e.g. Puestow *et al.*, 2001; Legleiter *et al.*, 2002; Marcus *et al.*, 2003). The highest accuracies were recorded for images of Helm Beck and the River Eamont, both of which were captured during June when water depths were at their lowest and the sun angle was highest. Lower accuracies were reported for imagery collected in February (River Caldew) and April (Trout Beck). Visual inspection of classification disparities suggested that the major sources of classification error occurred in areas of shadow which were misclassified as deep water; in areas of exposed but vegetated substrate, which were misclassified as shallow; at channel margins where it was difficult to distinguish between water and overhanging woody vegetation (prior to foliage emergence); and in areas of turbulent deep water, where white water resulted in a high level of reflectance and therefore classification as shallow water. A further source of error may be with the validation data themselves. As discussed, several researchers have suggested that classified, remotely sensed data may be more accurate than its comparative field data. A similar impact may also exist when using visual interpretation of images for validation. In this case it may be that subtle depths changes are simply undetectable by eye. This was felt to be particularly true for the validation of the Trout Beck image where it was considerably difficult to detect depth variations by eye. The ability to detect depth changes may depend on such factors as the underlying geology, substrate size, and image clarity. However, despite these errors, classification accuracies were considered to be acceptable. In addition to the main accuracy assessment, a small amount of field data measuring water depth was collected in relation to the River Caldew image. Here, approximately 50 depth measurements were taken (5 across each of 10 transects) during April 2004. Each measurement site was mapped directly on to a reproduced copy of the raw aerial photograph. This technique has previously been recommended by Marcus *et al.* (2003) as it reduces errors due to mismatches in georeferencing between photography and map data. Table 4.10 demonstrates that the various classes do exhibit varying depth characteristics, which was confirmed to be statistically significant by a Kruskal-Wallis analysis of variance. Due to the fact that field validation and photography were not undertaken on the same day, it is difficult to infer absolute depth by calibrating image interpretation against field measurement. However, measurements can be used to validate relative depth classifications of shallow/moderate/deep.

Table 4.10: *Relationship between unsupervised classification categories and field-based depth measurements.*

Water depth class	Range in depth measured (cm)
Shallow	9-25
Moderate	19-60
Deep	54->80

4.3.3 Classification of mosaicked images

The above classifications were based on individual images or segments of images at a small number of discrete locations. However, to be applicable at the catchment-scale, it was important to assess whether entire mosaics could be analysed in one go. The unsupervised classification procedure described above was therefore applied to a selection of aerial mosaics. Unfortunately, a major limitation was encountered at this stage of the analysis. As Figure 4.9 shows, changes in scene and illumination variations throughout the mosaic severely hinder the ability to use automated classification procedures. Segments of the mosaic with a higher base illumination may be incorrectly classified as shallower than those with a lower base illumination. In terms of addressing the issue of variable illumination, the standard procedure is to apply image pre-processing techniques such as colour balancing and histogram equalisation. The theory behind these techniques is to match the histogram of one image to that of its neighbour.

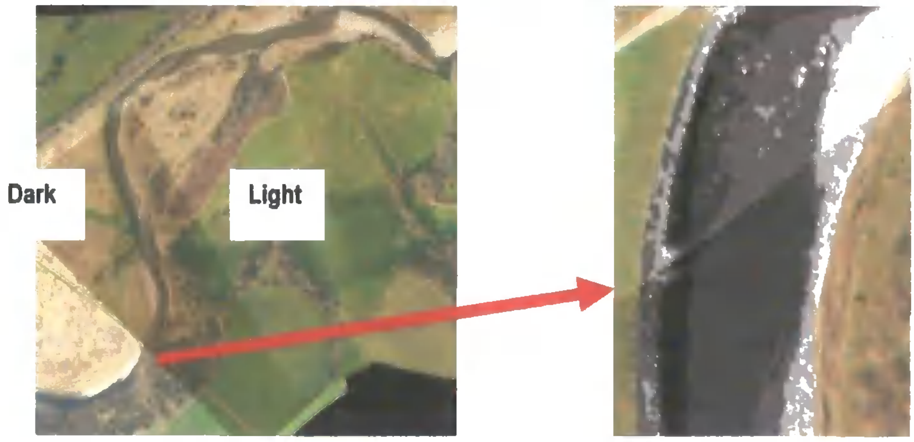


Figure 4.9: *Illumination variations between images lead to error in the classification of water depth.*

Whilst this may provide visually pleasing results by smoothing out local differences in illumination, it can introduce severe errors into the assessment of bathymetric variables as a result of changes in scene (Carbonneau *et al.*, 2004). For example, two images in close proximity may have very different spectral histograms due to a difference in feature composition, something which is common in highly variable fluvial environments (Figure 4.10). For this reason colour balancing was not applied to the imagery.

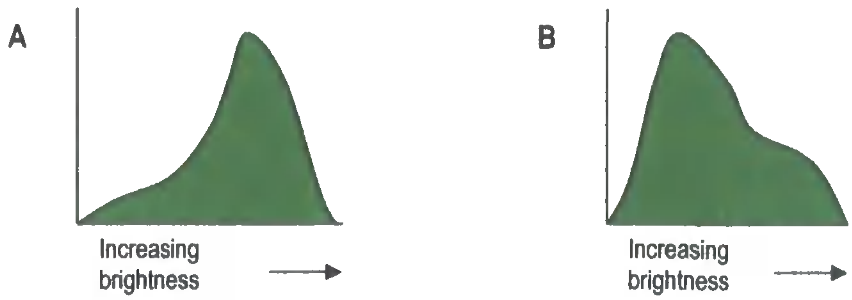


Figure 4.10: *Variation in the spectral histogram of two theoretical images due to variations in scene. Both images are assumed to have the same base illumination. (a) An image containing a high proportion of exposed substrate may have a histogram which tends to the right (b) An image with a high proportion of deep water may have a histogram which tends to the left. (Based on Carbonneau et al., 2004).*

A method for addressing this issue has been developed by Carbonneau *et al.* (2004) using 1-3cm resolution data. They suggest that if the initial brightness levels (I_{in}) can be determined, then images can be automatically corrected for illumination conditions. This they reason can be obtained if one conceptually removes the water medium to look at the brightness level of the bed, by identifying wetted substrate that is not submerged, located at the wet/dry interface. By comparing the spectral signature of wetted substrate to that in the next image it is possible to automatically correct for illumination. Unfortunately, this approach was not considered applicable to the case of the Eden catchment. Many of the images do not contain any areas of exposed wetted substrate and, even where they do, the pixel resolution is typically too coarse to capture their spectral signature without pixel mixing between dry/wetted and submerged clasts. The best solution identified was therefore to clip each mosaic into its constituent images, run unsupervised classification and undertake image interpretation separately for each image and then re-mosaic the classified images using the 'Merge' function available with ArcGIS (Figure 4.11). However, the time involved in doing this was considerable (approximately 2-3 mosaics per day), and due to time constraints within the scope of this project, the technique was judged unfeasible at the catchment-scale.

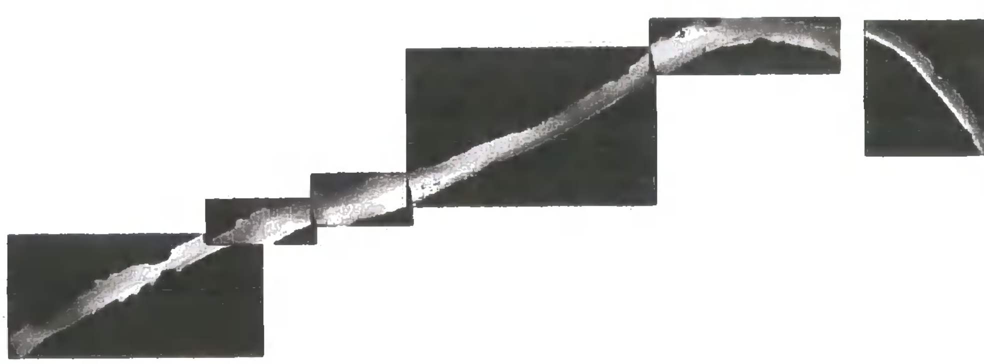


Figure 4.11: *Segmented mosaic re-merged following unsupervised classification*

4.3.4 Evaluation and comparison with the traditional walkover methodology

20cm digital aerial photography was not considered a viable alternative to the traditional walkover survey for mapping in-stream habitat at a catchment-scale, primarily due to the difficulties encountered as a result of varying illumination between images. However, as technology advances and higher resolution data becomes more commercially available and cost effective, dealing with the issue of illumination variation should be possible using techniques such as those developed by Carbonneau *et al.* (2004). If these issues can be overcome, then the application of remote sensing to in-stream habitat mapping will become very promising and exciting as high levels of classification accuracy were reported for individual images. As discussed, in much of the literature surrounding this topic, habitat maps produced using remote sensing techniques may represent a considerable improvement in the ability to capture and quantify spatial heterogeneity in stream habitat when compared with the traditional polygon mapping techniques employed in ground reconnaissance surveys.

In terms of classifying salmonid habitat from remotely sensed data, Figures 4.12 and 4.13 compare the unsupervised classification of relative water depth with the field mapping of salmonid habitat undertaken by APEM Ltd for a 200m reach of the River Eamont in 2004. The results are encouraging, showing that there is a clear correspondence between areas classified as deep and shallow and those habitat types that would be expected to have deep or shallow water. For example, fry habitat was located in areas constituting a high degree of shallow water whilst parr habitat was located in areas predominately classified as moderate depth. The simple classification into shallow/moderate/deep water is not sufficient to distinguish between riffle and fry habitat. This may be improved by using a larger number of classes. Alternatively, an expert fisheries biologist viewing the imagery may be able to identify areas of likely fry habitat based on their geomorphological context. As shown, fry habitat is located at the upstream end of the shallow/riffle area, as would be expected based on knowledge of habitat utilisation. At the transition from pool to riffle, water velocities are increasing and the difference in hydraulic head between the upstream pool and downstream riffle causes water to be drawn down into the streambed (the hyporheic zone) and interstices between gravels where fry are found (Summers *et al.*, 1996). Using this knowledge it may be possible to estimate those areas of shallow riffle that are likely to represent optimal fry habitat. The addition of data regarding substrate size would also greatly improve classification into salmonid habitat type. Again, as remote sensing technology advances it may become possible to classify substrate for large areas automatically (e.g. Carbonneau *et al.*, 2005).

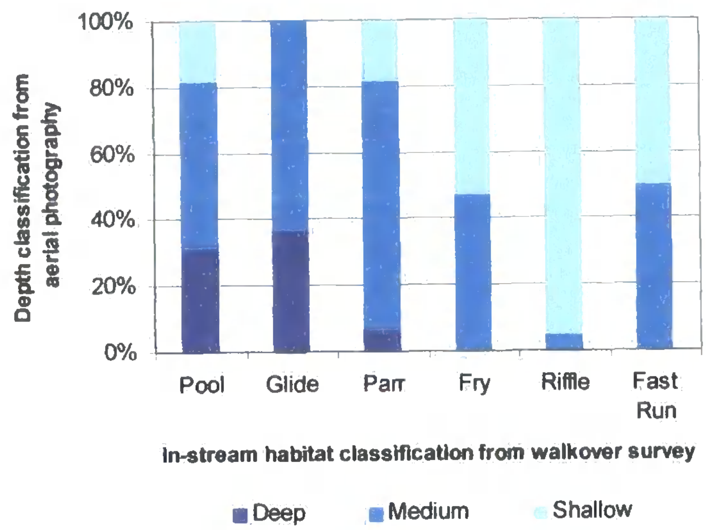


Figure 4.12: *The relative depth composition (from aerial photography)of in-stream habitat types mapped during a walkover survey of the River Eamont by APEM Ltd. The data presented is for a 200m reach.*

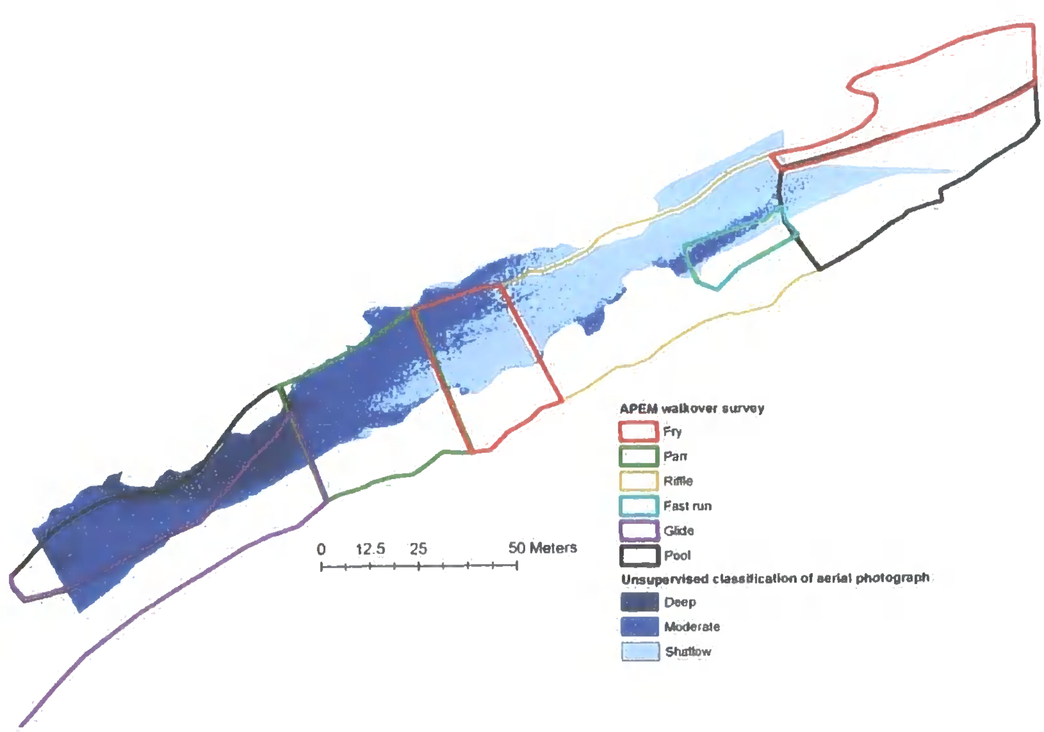


Figure 4.13: *Visual comparison between relative depth classification from aerial photography and in-stream habitat classification from a walkover survey of the River Eamont by APEM Ltd.*

4.4 Channel slope and biotope analysis

Due to the difficulties encountered in classifying water depth and in-stream flow type using automated image processing, a second methodology has been devised to try and extract this information from alternative remotely sensed data. At low to moderate flows hydraulic variables such as velocity and depth are intrinsically related to channel slope (Sear *et al.*, 2003). It has therefore been assumed that this also applies to flow type and biotope. If this is the case, it may be possible to gain an estimate of in-stream habitat diversity from information regarding channel slope. Research by Padmore (1998) has promoted the use of physical biotopes (e.g. waterfall, spill, cascade, rapid, riffle, run, boil, glide and pool) as the fundamental component of in-stream habitat. These, Padmore states (p.25), "can be identified by dominant flow type [e.g. Table 4.11], as a particular combination of substrate and hydraulic parameters will have a characteristic surface flow type". Biotopes are considered to provide an ecologically relevant classification of in-stream habitat, and as discussed in Chapter Two, juvenile salmonids are particularly associated with the riffle biotope.

Table 4.11: *Descriptions of flow types used to identify biotopes in the field (reproduced from Padmore, 1998).*

Flow type	Description	Associated biotope(s)
Free fall	Water falls vertically and without obstruction from a distinct feature, generally more than 1m high and often across the full channel width	Waterfall
Chute	Fast, smooth boundary turbulent flow over boulders or bedrock. Flow is in contact with the substrate, and exhibits upstream convergence and downstream divergence.	Spill – chute flow over areas of exposed bedrock Cascade - chute flow over individual boulders
Broken standing wave	White-water 'tumbling' waves with crest facing in an upstream direction. Associated with 'surging' flow.	Cascade – at the downstream side of the boulder flow diverges or 'breaks' Rapid
Unbroken standing wave	Undular standing waves in which the crest faces upstream without 'breaking'.	Riffle
Rippled	Surface turbulence does not produce waves, but symmetrical ripples which move in a general downstream direction.	Run
Upwelling	Secondary flow cells visible at the water surface by vertical 'boils' or circular horizontal eddies.	Boil
Smooth boundary turbulent	Flow in which relative roughness is sufficiently low that very little surface turbulence occurs. Very small turbulent flow cells are visible, reflections are distorted and surface foam moves in a downstream direction. A stick placed vertically into the flow creates an upstream facing 'V'.	Glide
Scarcely perceptible flow	Surface foam appears to be stationary and reflections are not distorted. A stick placed on the water surface will remain still.	Pool – occupy the full channel width. Marginal deadwater – does not occupy the full channel width.

The aim here is to evaluate whether channel slope and therefore biotope can be extracted remotely using the NEXTMap Great Britain ™ 5m DTM and GIS processing. Previous studies have attempted to extract channel slope from digital topographic data but have commented that the resolution of data available (OS Panorama 50m DEM) was too coarse (Coley, 2003). The ratio between DTM horizontal and vertical resolution can have a significant effect on slope which is calculated as the difference in elevation between adjacent cells divided by the difference between them. Previous studies have also utilised DTMs in integer format restricting calculations to a limited number of discrete values (Coley, 2003) The NEXTMap DTM represents a significant advance in both horizontal and vertical resolution, and is provided in floating point format increasing the number of possible elevation values. The aim here is to assess whether these improvements are significant enough to allow the accurate calculation of channel slope.

4.4.1 Methodology for the calculation of channel slope

The following steps were taken to derive a raster layer of channel slope (herein called 'DTM slope') for the Eden catchment using ArcGIS:

(1) First the DTM (`DTMoriginal`) was pre-processed to remove or fill any sinks present in the data. A sink is a cell or group of cells which have a lower elevation than all the surrounding cells and cannot therefore be assigned an outward slope or flow direction. Sinks are considered to most commonly occur due to errors in the raw data and the procedure of filling sinks has become a standard pre-processing step commonly undertaken within GIS hydrological analyses (Coley, 2003). However, it should be noted that sinks may be naturally occurring features within the landscape, particularly, in glacial or limestone environments, and there are some suggestions within the literature that sinks should not be filled. The identification and removal of sinks was undertaken in this case, as the hydrological analysis tools used to derive the channel network required a depressionless DTM. This was achieved using the "Fill Sinks" tool available within the ArcGIS Hydrological Analysis add in, downloaded from the ESRI website (www.esri.com). The output file was named `HydroDTM`.

(2) A drainage network for the catchment was then derived from `HydroDTM` using the ArcGIS Hydrological Analysis add in. This calculated flow pathways throughout the catchment using a single flow routing, D8 algorithm. First, a flow direction grid was created, where water was assumed to flow from one cell to the next following the line of steepest descent. This was

then used to produce a flow accumulation grid by counting the number of cells upstream of each point in the network. Finally, the flow accumulation grid was thresholded to define the stream network. In this case a threshold of 40,000 cells, equivalent to an upslope contributing area of 1km², was selected. The stream network was output as an ESRI GRID file (`StreamNet`) with 5m spatial resolution; cells above the threshold value were given a value of 1 whilst those below were set to 'No Data'. It is now generally agreed that the use of multiple flow routing algorithms (e.g. D∞) provide a more realistic model of flow pathway. Unfortunately, at a DTM resolution of 5m, the computational demands of calculating a flow accumulation grid for the Eden catchment using D∞ were too high compared with the computing power available. Maintaining a 5m resolution was considered critical and the D8 algorithm was instead selected to reduce computational demands. Visual comparison was made between the channel position determined from; (1) the 5m DTM and D8 algorithm; (2) a re-sampled 20m DTM and D∞ algorithm; and (3) OS LandLine® data. This showed that, in general, there was very little difference between the two flow routing algorithms and in terms of channel position the single routing algorithm frequently produced a more accurate representation of the channel as a result of the greater DTM resolution.

(3) Elevation values from the original raw DTM (prior to the filling of sinks) were then extracted for the drainage network cells using the Raster Calculator and the following Map Algebra equation:

`StreamNet*DTMoriginal` [4.7]

All surrounding cells were set to "No Data". This forces the calculation to determine the channel slope as opposed to that of the valley sides (Coley, 2003).

(4) The percentage slope was then calculated using the Slope algorithm available within the ArcGIS Spatial Analyst extension. This uses a moving 3x3 mask to calculate slope for the centre cell from each of its 8 neighbours (Figure 4.14). The equation used to calculate the percentage slope is (Dunn and Hickey, 1998):

$$S = \left(\sqrt{(S^2_{e\text{-}w} + S^2_{n\text{-}s})}\right)*100 \qquad [4.8]$$

East-West slope is given by:

$$S_{e\text{-}w} = \frac{(z_3 + 2z_4 + z_5) - (z_1 + 2z_8 + z_7)}{4*2*d} \qquad [4.9]$$

North-South slope is given by:

$$S_{n\text{-}s} = \frac{(z_1 + 2z_2 + z_3) - (z_7 + 2z_6 + z_5)}{4*2*d} \qquad [4.10]$$

Where: S = slope ratio in percent
z_1 to z_9 = elevations of cells 1 to 9
d = cell resolution

1	2	3
8	9	4
7	6	5

Figure 4.14: *Moving 3x3 grid used to calculate slope within ArcGIS.*

Based on a cell resolution of 5m the resulting slope values can be said to be calculated over a channel reach of approximately 15m, although this will vary slightly where the flow path follows a diagonal route. The slope values are non-directional.

4.4.2 Validation of channel slope

For validation purposes, channel slope as a percentage was measured in the field under low flow conditions using a level and staff at 111 sites across the catchment. As in the GIS, slope was measured over a scale of 15m. Readings were taken at 3m intervals throughout the selected reach and integrated in Microsoft Excel to give the slope over 15m (Figure 4.15). This 5 measurement procedure was adopted to minimise error due to bed roughness. For example, placing the staff on a large cobble could introduce error in the order of 0.05-0.25m in level readings. Water surface slope was also calculated by measuring the depth at each record point through the reach and subtracting this from the actual level reading. The 111 validation sites were distributed across a range of channel slopes and biotopes, selected using random stratified sampling, to enable evaluation of the relationship between channel slope and biotope. Each 15m reach was selected so that it fell completely within one dominant biotope. The biotopes used in this study were cascade, step pool, rapid, riffle, run and glide/pool. This classification does deviate slightly from that presented by Padmore (1998) as waterfalls and boils are excluded, and the spill/cascade category merged. An additional category of step pool was also included to represent the commonly occurring channel reaches of cascade (step) and pool sequences where pool spacing is less than 15m. The 111 slope validation sites were then plotted in a new ArcGIS point shapefile `SlopeValidation`, with the help of aerial photography and GPS readings taken at the time of data collection. The corresponding DTM slope values were extracted to this new file using the 'Extract to Point' function available in Spatial Analyst. The file was then exported to the statistical package SPSS v.12 for further analysis.

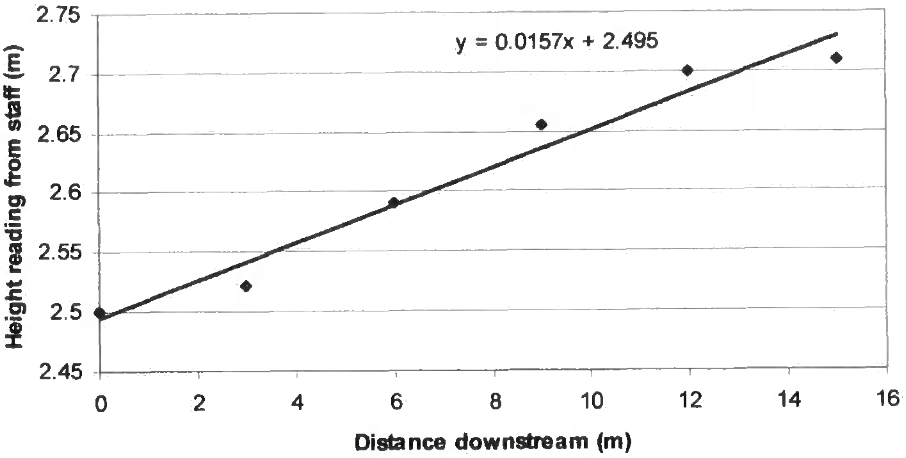


Figure 4.15: *Example of bed slope calculation for a riffle on Scandal Beck. Slope = 1.57%*

Prior to analysis it was necessary to apply a Natural Log+1 transformation to all three slope categories to approximate a normal distribution as confirmed by Kolmogorov-Smirnov tests. A constant of 1 was added to all sites due to the number of 0% DTM Slope sites. Linear regression analysis was then used to compare the DTM Slope values to the observed values, and, as presented in Table 4.12, a significant correspondence between the two was observed, with R^2 values of 61.9% and 60.8% for water surface slope and channel bed slope respectively. Histogram and normal probability plots of the residuals confirmed the assumption of normality in the error term. However, analysis did highlight the presence of a number of significant outliers (Figure 4.16) with standardised residuals greater than 2 standard deviations of the mean.

Table 4.12 *Regression analysis between observed channel slope and DTM slope (all sites included).*

Regression Equation	R^2 value	Standard error	Sig.
Ln(observed bed slope + 1) = 0.668(Ln(DTM slope + 1)) + 0.382	0.608	0.336	0.000
Ln(observed water surface slope + 1) = 0.743(Ln(DTM slope + 1)) + 0.205	0.619	0.387	0.000

Table 4.13 *Regression analysis between observed channel slope and DTM slope (significant outliers removed).*

Regression Equation	R^2 value	Standard error	Sig.
Ln(observed bed slope + 1) = 0.796(Ln(DTM slope + 1)) + 0.308	0.790	0.263	0.000
Ln(observed water surface slope + 1) = 0.864(Ln(DTM slope + 1)) + 0.113	0.776	0.301	0.000

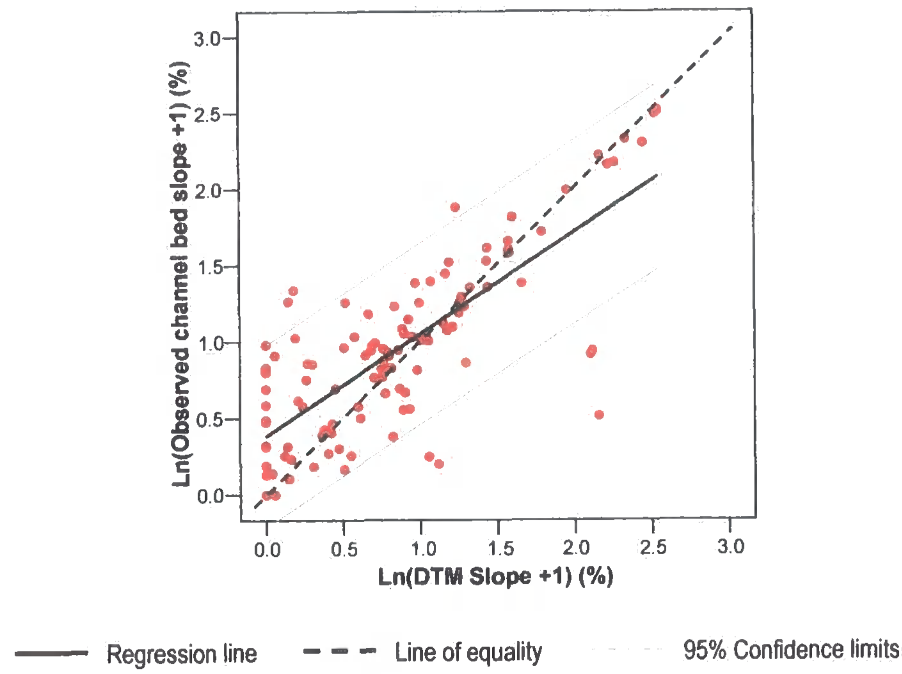


Figure 4.16: *Regression analysis between observed channel slope and that calculated from the DTM.*

On closer investigation a number of these outliers could be explained by two main factors. First, alteration to channel morphology, including channel planform, bed topography and ultimately channel slope which occurred during the January 2005 floods (post DTM data collection). This emphasises the fact that rivers and biotopes are dynamic features which will change position and form through time. This highlights the importance of considering the date of data capture when evaluating results and also raises the possibility of using digital topographic data and hydrological analysis as a means of monitoring channel change over time. Second, outliers corresponded with reaches of tree coverage. Although the DTM has been processed to remove the effects of trees and buildings some errors are still likely to remain where these features occur, as described in Chapter Three (Section 3.5.2). As such, the analysis of channel slope may be unsuitable in heavily forested catchments. Within the Eden catchment manual processing has been undertaken prior to biotope analysis to exclude sites within large wooded areas or where substantial flood alteration is known. Removing these known outliers from the regression analysis increased the level of agreement between the field measured and DTM slope values (Table 4.13). An increase in the gradient of the regression line towards a value of 1 was also observed, indicating that the agreement was nearing a 1:1 relationship. t-tests were performed on the data (with and without outliers) to test the null hypothesis that the regression line was not significantly different to the line of equality at any stage in the analysis (Table 4.14). In all cases the null hypothesis was accepted (p>0.1). An increase in the t-statistic, on removal of the outliers, also confirmed that this did increase the strength of agreement observed between the two lines.

Table 4.14: *T-tests between the derived channel slope regression lines and the line of equality*

	With outliers		Without outliers	
	Bed slope	**Water slope**	**Bed slope**	**Water slope**
t-statistic	-0.9881	-0.6641	-0.7757	-0.4518
***p*-value**	>0.1	>0.25	>0.25	>0.25

In both cases (with and without outliers) water surface slope is nearer to approximating a 1:1 relationship than channel bed slope. An explanation for this may be that in shallow water biotopes water surface slope is likely to approximate the channel bed slope, but, in pools channel bed and water surface slope are likely to deviate from each other, as illustrated in Figure 4.17.

Figure 4.15: *Variation between water surface slope and channel bed slope within pools*

Whilst water surface slope is likely to be relatively low when measured at any scale within a pool, channel bed slope may vary considerably, and be relatively high, particularly, if it is only measured across the entrance (+ve slope) or exit (-ve slope) of the pool. Measured over the entire length of a pool, water surface slope and channel bed slope should correspond with each other but this would only have been achieved at sites with a pool length of 15m. Attempts were made to centre slope measurements across the deepest part of pools, but, as pools are rarely symmetrical, deviations between water surface and channel bed slope still occurred. Additionally, whilst radar may penetrate water to shallow depths, in deep-water pool reaches it is more likely to capture the elevation of the water surface than the channel bed. The remaining outliers may be the result of either error in the DTM calculation of channel slope or error in the measurement of channel slope in the field. Uncertainty in the estimation of slope from elevation data is a function of the relative magnitude of uncertainty in elevation as compared to the spacing between elevations over which slope is being calculated as expressed by equation [4.11] (Lane *et al.*, 2003b);

$$\pm\delta_s = \frac{\sqrt{2}\delta_e}{d} \qquad [4.11]$$

where: δ_s is the uncertainty in slope
δ_e is the uncertainty in elevation
d is the distance over which slope is calculated

With an elevation uncertainty of 1m and slope measured over 15m this equates to a value of uncertainty in slope of ±9.43%. However, despite these relatively high levels of uncertainty DTM slope was still observed to provide good representation of actual channel slope and investigation was extended to biotope analysis.

4.4.3 Methodology for biotope analysis

It has been assumed that physical biotopes are intrinsically related to channel slope at low to moderate flow. Preliminary analysis, plotting boxplots of slope distribution (both channel bed and water surface) for each category, supports this assumption (Figure 4.18). The boxplots again highlight the issue of measuring channel bed slope within pools. Further analysis has therefore focused on water surface slope alone. ANOVA confirmed that there was a statistically significant difference between the mean Ln(slope+1) of different biotopes and investigation was therefore extended using regression analysis to model the relationship between channel bed slope and physical biotope in more detail.

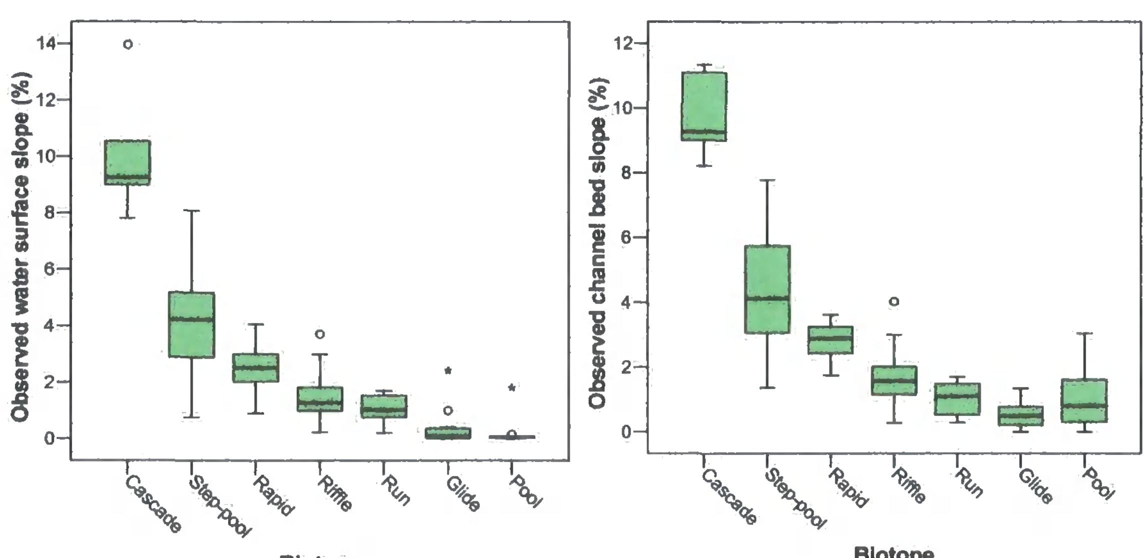


Figure 4.18: *Boxplots of the relationship between biotope and slope (a) water surface slope and (b) channel bed slope.*

Ordinal regression procedures have been followed due to the ordinal classification of biotope from high slope (cascade) to low slope (pool). Three separate regression models, complementary log-log, cauchit (inverse Cauchy) and logit were developed from the water surface slope measurements and biotope observations made in the field. The form of the regression equation [4.12] for each of the three models is shown below:

$$Link\,(\gamma_{ij}) = \theta_j - [\beta x_i] \qquad [4.12]$$

where:	*Link ( )*	is the link function
	γ_{ij}	is the cumulative probability of the j^{th} category for the i^{th} case
	θ_j	is the threshold for the j^{th} category
	β	is the regression co-efficient
	x_i	is the predictor value for the i^{th} case

Link functions:	Logit	$\log(\gamma/1-\gamma)$
	Log-log	$\log(-\log(1-\gamma))$
	Cauchit	$\tan(\pi(\gamma-0.5))$

Ordinal regression predicts the probability that a particular value of channel slope will belong to a particular biotope. These probabilities were then used to predict biotope from channel slopes measured in the field and calculated by the DTM and validation of the results is discussed in the next section.

4.4.4 Validation of biotope analysis

Formal accuracy assessment procedures were again followed to evaluate each model's ability to predict biotope and to assess the degree to which using remotely sensed data instead of field data affected biotope classification accuracy (Table 4.15). In addition to the statistics used previously, the Z-statistic [4.13] was also calculated as a measure of the statistical significance of the Kappa values reported for each model. The denominator is the approximate large sample variance of Kappa. At the 95% confidence level, the Z-statistic is significant if it is greater than 1.96, indicating that the classification is better than chance:

$$Z = \sqrt{\hat{\mathrm{var}}\,\hat{\mathrm{K}}} \qquad [4.13]$$

The Kappa statistic can also be used to compare two different error matrices, through a measure of the Z-statistic that describes whether the two error matrices are significantly different from each other [4.14]. Let $\hat{K}_1$ denote Kappa obtained from error matrix 1 and $\hat{K}_2$ denote Kappa obtained from error matrix 2. The Z test statistic is calculated from:

$$Z = \frac{\left|\hat{K}_1 - \hat{K}_2\right|}{\sqrt{\hat{\mathrm{var}}(\hat{K}_1) + \hat{\mathrm{var}}(\hat{K}_2)}} \qquad [4.14]$$

If the Z-statistic is greater than 1.96 the error matrices are significantly different; if it is less than 1.96 there is no significant difference at the 95% confidence level.

Table 4.15: *Accuracy assessment results for the estimation of biotope from observed and DTM water surface slope.*

Model	Observed water surface slope			DTM slope			Z statistic†
	Overall accuracy (%)	Kappa	Z statistic	Overall accuracy (%)	Kappa	Z statistic	(Pairwise comparison of observed and DTM error matrices)
Log-log	61.4	0.462	8.22	52.2	0.327	5.76	1.691
Cauchit	61.4	0.488	8.36	55.0	0.411	6.97	0.931
Logit	60.5	0.467	8.22	55.0	0.396	6.81	0.873

† At the 95% confidence level the Z statistic is significant if it is greater than 1.96.

Analysis indicated that for both observed water surface slope and DTM slope, the cauchit ordinal regression model resulted in the highest level of agreement between predicted class and observed biotope, with a Kappa value of 0.488 (observed slope) and 0.411 (DTM slope). The individual Z-statistic for all three models was highly significant indicating that the classifications were all better than chance. When applied to DTM slope values, the level of agreement between predicted and observed biotope was only degraded slightly, and analysis of the pairwise Z-statistic confirmed that for each model there was no significant difference between the error matrices developed for observed slope and those developed for DTM slope. This suggests that the use of remotely sensed data instead of field-based methods does not significantly limit the assessment of biotope biodiversity based on channel slope. However, not all biotopes were predicted by the regression models. In particular, the pool biotope failed to be identified by any model, whilst the run category was not predicted by the log-log and logit models and the rapid category was not predicted by the log-log model. Independent-Sample t-tests were applied to assess the null hypothesis that the mean Ln(slope+1) of each individual biotope was not significantly different to every other individual biotope (Table 4.16). The null hypothesis was rejected in all cases except between the step-pool/rapid, riffle/run and glide/pool categories, confirming that it may not be possible to distinguish between these categories using water surface slope alone.

Table 4.16 *Comparison of the mean Ln(water surface slope + 1) of each biotope using independent-sample t-tests. Figures shown in bold are not significant at the 95% confidence level.*

	Cascade	Step-pool	Rapid	Riffle	Run	Glide	Pool
Cascade		0.001	0.000	0.000	0.000	0.000	0.000
Step-pool	0.001		**0.085**	0.000	0.000	0.000	0.000
Rapid	0.000	**0.085**		0.001	0.000	0.000	0.000
Riffle	0.000	0.000	0.001		**0.076**	0.000	0.000
Run	0.000	0.000	0.000	**0.076**		0.000	0.000
Glide	0.000	0.000	0.000	0.000	0.000		**0.663**
Pool	0.000	0.000	0.000	0.000	0.000	**0.663**	

As a result these categories were merged and the regression models re-applied. As Table 4.17 illustrates this resulted in a significant increase in the level of agreement between predicted and observed biotope, with all models (excepting the log-log model applied to DTM slope) achieving a good agreement (Altman, 1981). The cauchit model again achieved the highest level of accuracy when applied to DTM slope and the Z-statistic confirmed that there was no significant difference between the observed and DTM slope error matrices.

Table 4.17: *Accuracy assessment results for the estimation of biotope from observed and DTM water surface slope following the merger of several classes.*

Model	Slope	Overall Accuracy (%)	Kappa	Z statistic (Pairwise comparison of observed and DTM error matrices)	Conditional Kappa			
					Cascade	Step-pool /rapid	Riffle/run	Glide/ pool
Cauchit	Observed	79.8	0.689	0.66	1.000	0.655	0.602	0.757
	DTM	76.1	0.631		0.650	0.834	0.547	0.625
Log-log	Observed	81.6	0.709	1.81	1.000	0.793	0.561	0.828
	DTM	71.5	0.546		0.650	0.862	0.403	0.634
Logit	Observed	78.8	0.675	0.65	0.790	0.635	0.602	0.757
	DTM	75.2	0.618		0.650	0.834	0.539	0.595

In terms of identifying individual biotopes all models achieved a moderate or greater level of agreement (Altman, 1981) between predicted and observed biotope. In all cases the riffle/run category reported the lowest level of agreement which was disappointing due to its importance as juvenile salmonid habitat. However, one of the advantages of using a continuous variable and ordinal regression is that it facilitates fuzzy classification of biotopes. This recognises the transition zone between biotopes based on the probability of class membership, for example, reaches that are definitely riffles, reaches that are either riffles or rapids and those that are definitely rapids. This is demonstrated in Figure 4.19 which maps the probability that riffle/run habitat exists for one tributary of the river Eden. Recognising these transition zones between biotopes is important as it is these reaches which are likely to be most sensitive to changing discharge or flow regime.

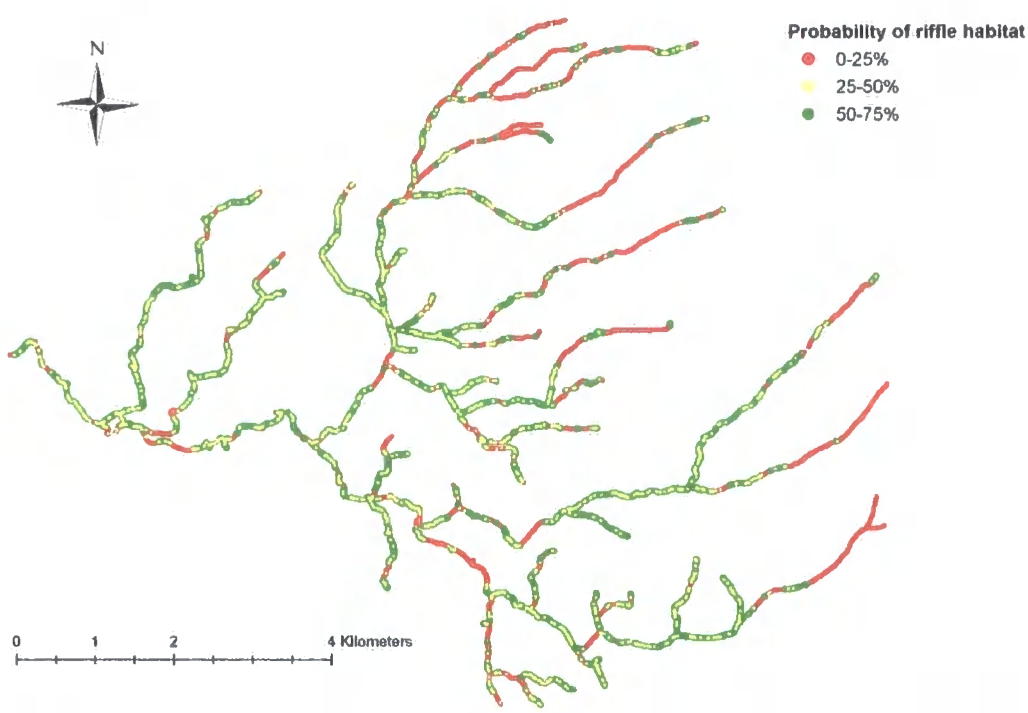


Figure 4.19: *Probability that in-stream habitat is of the riffle/run type in Trout Beck, a tributary of the River Eden.*

4.4.5 Comparison of DTM based channel slope and biotope analysis with the traditional walkover methodology

In-stream habitat availability and diversity are typically assessed through the conventional mapping or recording of flow types and physical biotopes in the field during walkover surveys. However, using a surrogate variable, such as DTM-based slope analysis, has a number of advantages for evaluating in-stream habitat availability. First, channel slope is an objective variable that can be physically measured and which remains relatively stationary under differing flow conditions. Conversely, biotope mapping in the field is reliant upon the ability of the individual surveyor to (subjectively) identify and classify channel reaches as a particular biotope dependent upon the flow conditions at that point in time. Research by Maddock *et al.* (2006) analysed observer variability in identifying habitat units for four different field based mapping techniques, finding variability between observers to be considerable especially when the spatial distribution of units was considered and not just the total composition of a reach. A second difficulty in mapping biotopes in the field, is that they do not exhibit distinct boundaries between types, yet surveyors are required to map them as discrete units. Instead, using a surrogate, continuous variable such as slope has the advantage that it can detect and record this gentle transition from one biotope to the next. This is particularly true if the aforementioned fuzzy classification of biotopes is applied. Similarly, use of a continuous variable enables variation within the same class to be identified, for example, high-gradient riffles compared to low-gradient ones, something which is far more difficult to classify by eye. Third, the DTM slope analysis produced a slope value and hence biotope classification every 5m downstream thereby recording in-stream diversity at a finer scale than most walkover surveys that will typically record a dominant flow type through an entire reach or the number of flow types present with no record as to their proportions or distribution. For example, one 100m pool followed by one 100m riffle is likely to have different ecological implications to five 20m pools interspersed with five 20m riffles.

However, there is a high level of uncertainty (±9.43%) associated with the determination of channel slope and hence biotope from remotely sensed DTM data. Therefore, as a technique it may be more suited to the broad catchment-wide assessment of habitat availability and diversity, than the detailed analysis of in-stream habitat pattern within a particular reach, as would be required when identifying specific reaches for individual restoration projects. The range of channel widths over which the technique is applicable may also be restricted by DTM resolution. Assuming an approximate pool spacing of 5-7 times the channel width (Sear *et al.*, 2003) and a 5m resolution DTM, the optimal channel width for applying this technique probably lies in the range of 3-10m wide, although further research would be required to clarify this. Errors were also identified within wooded reaches and caution should therefore be taken if applying the technique in heavily forested catchments. A further source of error was observed in reaches which had undergone morphological change during the January 2005 floods and the date of data capture should therefore be considered when analysing results. However, despite the uncertainty and sources of error, results showed that channel slope within the Eden catchment could be calculated with a good degree of accuracy from the NEXTMap Britain ™ DTM and that there was little loss in accuracy by inferring biotope using DTM Slope compared with water surface slope measured in the field.

4.5 Chapter summary

The aim of this chapter was to identify and develop tools for the derivation of salmonid relevant habitat data at the in-stream and riparian scale using 20cm digital aerial photography and the NEXTMap Great Britain ™ 5m digital terrain model (DTM). Three different methodologies were evaluated, both in terms of their ability to accurately classify habitat, and in terms of their practicality compared with traditional ground reconnaissance survey techniques. Based on the results of this analysis Table 4.18 presents a summary of those variables which can and cannot be quantified at the catchment-scale using the remotely sensed data and three methodologies applied here.

Table 4.18: *The potential for catchment-scale assessment of in-stream and riparian habitat controls using remotely sensed data.*

X – the variable can be quantified effectively at the catchment-scale;

O – the variable can be quantified but not cost or time effectively at the catchment-scale

Habitat control	Walkover	Visual assessment of aerial photography (20 cm resolution)	Automated classification of aerial photography (20 cm resolution	GIS processing of digital topographic data (5m resolution)
In-stream scale				
Water depth	O		O	
Physical biotope	X			X
Substrate (in detail e.g. distinguish gravel from pebble)	X			
Substrate (broad classes e.g. bedrock, coarse, fine)	X	X		
Channel slope	O			X
Riparian-scale				
Bank erosion presence	X	X		
Bank erosion cause	X	X		
Stock access	X	X		
Percentage overhead cover	X	X		
Tree density	X	X		
Land cover	X	X		
Bank modification	X			
Depositional features	X			

Each of the methodologies is summarised in turn. First, visual assessment of aerial photography was used to map riparian condition and in-stream features detectable by eye. Results were extremely promising, particularly, for the mapping of bank erosion and channel cover, both of which are considered to strongly impact the quality of salmonid habitat. The technique was relatively cheap and rapid to apply at the catchment-scale and is considered an excellent alternative to ground surveying in large catchments. Second, automated image processing techniques were used to extract relative depth information from aerial photography. Unfortunately, whilst the accuracy of classifications for individual images was extremely high, at the current time, this technique was not considered applicable at the catchment-scale, primarily as a result of the illumination variations between images. Third, DTM-based channel slope analysis was used to estimate in-stream biotope type. Results suggested that the use of remotely determined channel slope compared with that measured in the field had little effect on the accuracy of biotope classifications generated, and that this is a potential technique for the general characterisation of in-stream flow conditions and salmonid habitat suitability. The catchment-wide habitat data collated by these techniques is related to salmonid population data in Chapter Six of this thesis to investigate the hypotheses identified in Chapter Two regarding habitat controls on salmonids.

Chapter Five - The impact of hydrological connectivity on salmonid populations at the catchment-scale

5.1 Introduction

Chapter Four focused on the identification and validation of tools for the assessment and quantification of riparian and in-stream salmonid habitat controls. However, as discussed in Chapter Two (Section 2.7.3), it is increasingly recognised that controls upon salmonid habitat may also extend to landscape factors and processes, such as geology, soil type, altitude, climate, relief and most notably catchment land use and land management (e.g. Johnson and Gage, 1997; Allan and Johnson, 1997; Hendry *et al.*, 2003). These factors are assumed to influence the delivery of water, sediment, solutes and organic matter to the channel network and hence impact local in-stream habitat and ecology. A different suite of tools are required to quantify these distributed landscape variables compared with those utilised in Chapter Four for the more localised assessment of riparian and in-stream controls. To this end, technological advances in computing power, GPS, GIS and remote sensing have enabled such tools to be developed. As discussed in Chapter Three (Section 3.1), there is a rapidly growing range of regional and national databases providing information on landscape variables, many in digital format (Johnson and Gage, 1997). In line with this proliferation of data sources has been a rise in ecological models including landscape variables within their parameter suite, but to date there has been little agreement as to the ecological significance of such variables (e.g. Wang *et al.*, 2003 compared with Stauffer *et al.*, 2000). In-stream and riparian habitat controls are intrinsically linked to salmonid habitat through their direct proximity with the channel. Landscape variable impacts differ as they are filtered by the degree of connection between them and the channel, and the ease with which material can be mobilised, transported and delivered through the catchment. This process is primarily achieved by running water and it is therefore the level of hydrological connection between the catchment and channel network that is important, potentially more so than the spatial distribution of land cover type (Burt, 2001). The nature (i.e. surface lateral flow, near surface lateral flow, groundwater) of hydrological connection has also been shown to influence the delivery of water, sediment, solutes and organic matter to the channel (Croke and Hairsine, 2006). It is also important to consider the way in which each connected land parcel contributes to the catchment-scale as a whole in the context of dilution effects. Studies of these processes have rarely been included in assessments of the impact of land cover and land management on ecology at a catchment-scale (Meador and Goldstein, 2003).

There is therefore a need to combine knowledge of the spatial distribution of landscape variables such as land use with knowledge of the spatial distribution of hydrological processes and relate this to spatial patterns of ecological performance (in this case salmonid abundance) at a catchment-scale, if the impact of landscape variables is to be truly examined. This may be achieved through the use of mathematical hydrological models linked to remotely sensed landscape variables, validated using catchment sources of ecological data (Lane *et al.*, 2006). With regard to selecting an appropriate model, Chapter One (Section 1.4.2) identified that such a model should be spatially distributed and explicitly incorporate treatment of hydrological connectivity (Burt, 2001; Lane *et al.*, 2006), but that it should also be parsimonious in terms of computational requirements if it is to be applicable at a catchment-scale. Lane *et al.* (2006) have developed such a model, The Sensitive Catchment Integrated Modelling Platform (*SCIMAP*) (www.*scimap*.org.uk) based upon the premise that surface and sub-surface lateral hydrological connectivity is primarily controlled by surface topography.

SCIMAP is an environmental model which assesses the relative risk of diffuse pollution in catchments within a probabilistic framework. It is based upon the conception of catchments as organising entities, within which a set of flow paths accumulate distributed sources of contaminants from across the landscape into receiving waters, where they may become a pollution problem (Lane *et al.*, 2006). In this way, it explicitly connects land units and sources of pollution through relevant hydrological delivery mechanisms integrated through to the channel. Analysis within *SCIMAP* follows a three-step procedure, according to the main controls over diffuse pollution generation within watercourses. First, the risk that contaminants may be generated and exported within the landscape is evaluated (e.g. Heathwaite *et al.*, 2000, 2003). However, as Lane *et al.* (2006, p243) state, "rather than casting the expected export as a volume of material produced (as in traditional modelling approaches) it is specified as a risk of material being produced, in relative terms compared with other units within the landscape". Second, the risk of contaminant delivery is determined based on catchment topography and the location of land units which will influence the dominant hydrological pathway and the risk of coupling (hydrological connection) along flow lines (e.g. Heathwaite *et al.*, 2000; Burt, 2001; Lane *et al.*, 2003a, 2004); and third, connected pollutant sources are accumulated along flow paths and integrated through to the drainage network, taking account of dilution potentials. Whilst absolute details of contaminant delivery are not predicted, these may not actually be necessary for management; rather it is the spatial distribution of risk, with one location compared relatively to another, which is necessary for targeting resources. *SCIMAP* is currently under development, and as yet is only developed to evaluate hydrological connectivity in terms of saturation-excess overland and shallow sub-surface flow. Therefore, at the time of this research, it is only suitable as a tool for assessing the risk of pollution from contaminants that are largely transported by surface pathways, such as fine sediment, phosphorus bound to fine sediment, and microbial risks. The availability of catchment-wide electrofishing data (Chapter 3, Section 3.3.) for the Eden catchment, coupled with the *SCIMAP* model presents a rare and exciting opportunity to examine the relationship between landscape hydrological connectivity and ecology, in this case salmonid abundance.

In accordance with Objective (2) of the thesis, the aim of this chapter is to evaluate and to validate the potential of the *SCIMAP* model as a tool for assessing and quantifying the impact of catchment-scale controls on salmonid habitat. In particular, the aim is to assess the role of hydrological connectivity in structuring the relationship between salmonid abundance and land cover at the catchment-scale. This will be achieved by relating the risk of (1) hydrological connectivity and (2) hydrological connectivity weighted by land cover, as determined through the *SCIMAP* model to catchment-wide data on salmon and trout fry abundance collected through semi-quantitative electrofishing. It has been hypothesised that surface hydrological connectivity weighted by land cover will influence salmonid abundance through its control over the delivery of fine sediment and nutrients to the channel. To evaluate this hypothesis and to examine further the functional significance of hydrological connectivity, analysis will then focus on the relationship between hydrological connectivity and the spatial pattern of gravel siltation and nutrient concentrations within the Eden catchment.

5.2 Methodology

A prototype of the *SCIMAP* model was provided by researchers from Durham and Lancaster Universities (Lane *et al.*, 2006). It has been developed to operate within SAGA GIS software and a schematic representation of the modelling framework is presented in Figure 5.1. The *SCIMAP* model has been applied to the whole Eden catchment producing two different risk layers:

1. The risk of surface hydrological connectivity between the channel network and catchment; and
2. The risk of surface hydrological connectivity weighted by land cover.

Statistical analysis within SPSS v.12 has then been undertaken relating the risk assessments to the spatial pattern of salmonid presence/absence, abundance, gravel siltation, and water chemistry.

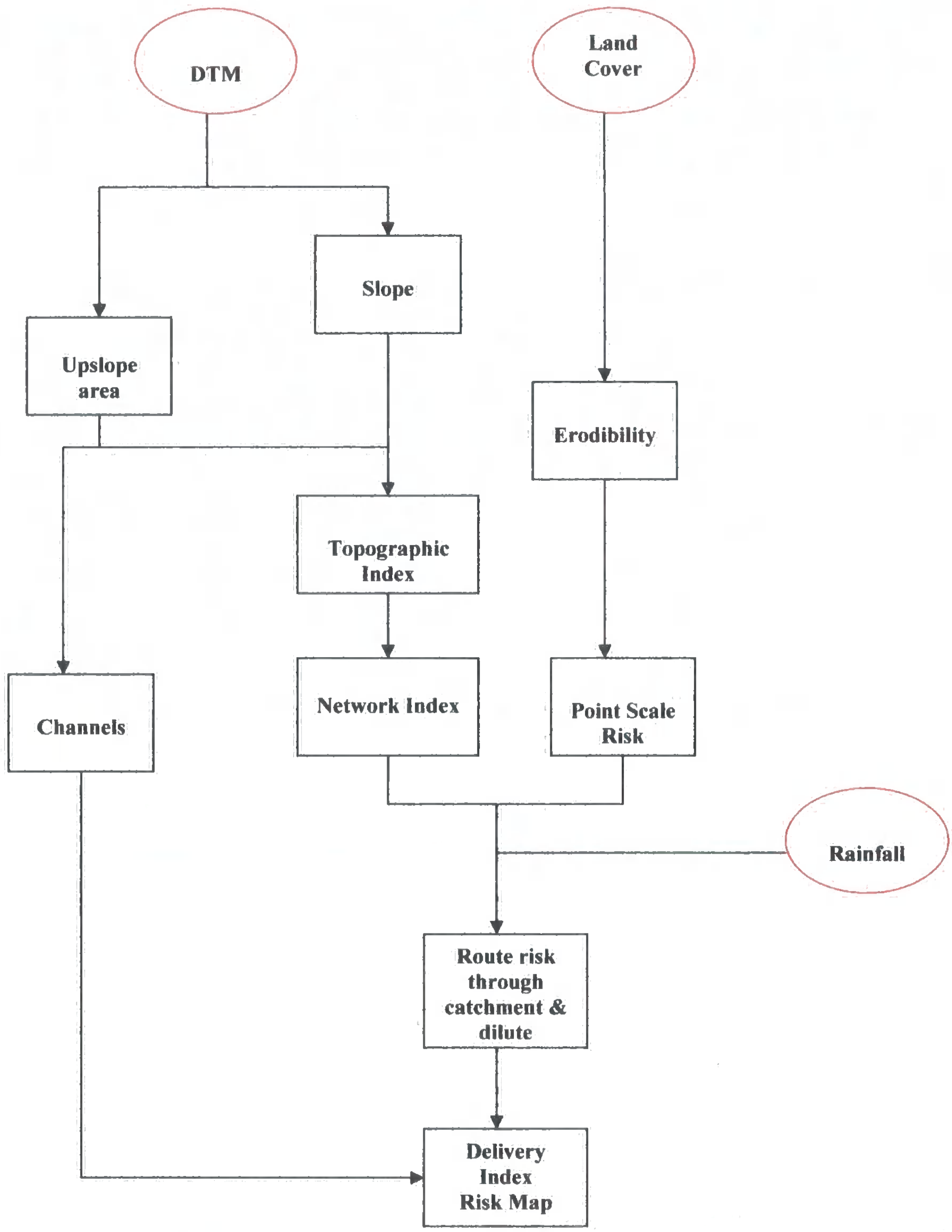


Figure 5.1: *Schematic representation of the SCIMAP model. (Red boxes indicate model inputs). (Adapted from www.scimap.org.uk)*

5.2.1 Application of the *SCIMAP* framework to the Eden catchment

The first stage in applying the *SCIMAP* model to the Eden catchment involved identifying the risk of pollution generation and export potential within the landscape. Within *SCIMAP* each land parcel at location *(i)* is assigned a risk of generation for the contaminant in question, parameterised by expert judgement based on landscape parameters (Lane *et al.*, 2006), in this case land cover. Land cover information was obtained from the Centre for Ecology and Hydrology's Land Cover Map 2000, Level 2 vector dataset, and has been used here as a proxy for agricultural activity (land use and land management) within the catchment. As discussed in Chapter Three (Section 3.6), whilst land cover may not always provide an exact indication of land use it is the best data available at the current time due to the confidentiality of the agricultural census. Land cover was specifically parameterised for its risk of soil erosion based on the expert judgement of the *SCIMAP* developers, scaled between 0 (low risk of erosion) and 1 (high risk of erosion) (Table 5.1). It was decided to concentrate on the risk of soil erosion and fine sediment generation due to the perceived negative impact on salmonid abundance as a result of fine sediment inputs into spawning redds (Soulsby *et al.*, 2001). Research has shown fine sediment delivery from catchment sources to primarily occur by overland flow or shallow sub-surface drain flow (Walling *et al.*, 2002) making it particularly suitable to assessment by the *SCIMAP* model which is currently developed to assess the risk of surface and shallow sub-surface hydrological connectivity. The dataset was interpolated in ArcGIS to a resolution of 20m and reclassified in line with Table 5.1 using the 'Reclass' function available within Spatial Analyst to produce a layer of relative erosion risk, which was then exported to SAGA GIS for analysis. Reaney, *et al.* (in preparation) have been exploring alternative parameterisations of these weightings using inverse modelling and sensitivity analysis methods.

Table 5.1: *Classification of land cover risk (Lane et al., 2006)*

Land Cover	Risk
Arable	1.00
Intensively managed grassland	0.30
Extensively managed grassland	0.15
Peat and bog	0.10
Heath and bracken	0.05
Woodland	0.05

The second stage involved determining the ability for risk within the landscape to connect to the channel network. This is termed the delivery index, and in the case of surface hydrological connectivity, the delivery pathway focused on is saturation-excess overland flow. To calculate the delivery index, *SCIMAP* initially establishes the topographic index [5.1] across the catchment surface. The topographic index is a measure of the propensity to saturation (overland flow generation) of points on the land surface and has been widely applied in many hydrological modelling studies since the development of TOPMODEL (Beven and Kirkby 1979)

$$\ln(a/\tan\beta) \qquad [5.1]$$

where: a is the rainfall weighted upslope contributing area; and β is the local topographic slope

However, not all areas of saturation will connect, nor deliver material to the channel network during a storm event (Lane *et al.*, 2004). For example, if surface flow reaches an area downslope that is not fully saturated, water will infiltrate into the soil. Where this occurs, material in transport will be deposited before reaching the channel network (Figure 5.2). It is only from fully connected, saturated areas that particulate matter is likely to be transported uninhibited to the channel network. Research tracing sediment sources, mobilisation and transport has shown that only a relatively small amount of the total sediment mobilised within catchments may be actually transported to the channel (Walling *et al.*, 2002).

Figure 5.2: *Deposition of sediment within the landscape. This surface pathway is not fully connected.*

To account for on-slope deposition *SCIMAP* adopts a modification of the classic topographic index (Beven and Kirkby, 1979) called the network index (Lane *et al.*, 2004).The network index is based upon the lowest value of the topographic index along a given flow path from the point of interest to the drainage network. Assuming, that the principal driver of surface and near surface lateral flow is surface topography, and that all other run-off influencing factors are homogenous (e.g. soil type and vegetation), the network index can be said to control the delivery of material along that flow path to the river. Points with a low network index require a greater amount of rainfall, or more rapid water table rise, to connect to the channel network than those with a high network index, and are therefore likely to connect less frequently (Lane *et al.*, in review). In other words, points in the landscape which have a low network index present a lower risk in terms of pollutant delivery than those with a high network index, assuming uniform land cover. *SCIMAP* then assesses the relative risk of connection at point *(i)* compared with the rest of the catchment by rescaling values of the network index using a probability density function based on the catchment-wide distribution of network index values. Values are scaled from 0 (low connection probability) to 1 (high connection probability). In reality, connectivity is a dynamic process. As saturated areas expand and contract throughout rainfall events, the degree of connectivity between areas of the landscape and the channel network will vary (Lane *et al.*, 2004). Whilst the *SCIMAP* analysis as applied here, is time integrated and does not explicitly model variations in connectivity through time, it does inherently account for them. The network index represents the average relative risk of connection of a particular location through time. Those areas with a high network index are more likely to connect more frequently and therefore represent a diffuse pollution risk for a greater proportion of time than those with a low network index. The 5m NEXTMap Britain ™ DTM (see Section 3.5.2) was used to derive the slope, topographic index, network index, upslope contributing area, flow routing and channel network for the Eden catchment. Whilst this provides topographic data at a 5m resolution, it is currently unfeasible to apply data at this level of detail within *SCIMAP* for a catchment as large as the Eden, due to problems with file size and computer memory. Alternatively, the data were re-sampled to produce a 20m resolution DTM for the catchment. Whilst this is not ideal, it still represents topographic data at the sub-field scale and is a significant improvement in detail when compared with previously available datasets such as the 50m OS Panorama DEM.

The risk layer of surface hydrological connectivity (network index) was then convolved in *SCIMAP* with the former layer of erosion risk to produce a map of locational risk across the catchment that was accumulated and routed (using the $D\infty$ algorithm, Tarboton, 1997) through to the channel network to estimate the delivery 'loading' at each point in the channel network. The output channel network, to which risk was integrated, was defined by setting a threshold for the upslope contributing area above which cells were classed as part of the channel network. Following several trial runs a threshold of $1km^2$ was chosen for the upslope contributing area representing a balance between the level of detail required for comparison with fisheries data and the visual clarity of output risk maps at the catchment-scale. The delivery index (a probabilistic surrogate for pollutant concentration) was then calculated by *SCIMAP* by dividing the loading by the rainfall-weighted upslope contributing area. This accounts for the dilution potential of receiving waters (Figure 5.3). For example, a small stream with low discharge will require a lower delivery loading to present a pollution problem than a large river with a high discharge. Within this research, a catchment-scale layer of rainfall distribution was produced using data from the UK Meteorological Office and UK Climate Impacts Programme 2002 (UKCIP02) (see Section 3.7).

Figure 5.3: *Source integration and the dilution potential of receiving waters. Risk from across the catchment is accumulated, routed and integrated through to the channel network.*

The output risk layers were exported from SAGA GIS into ArcGIS GRIDs for further analysis due to the advanced visualisation and analysis capabilities available. It is important to note that classic model calibration was not required as *SCIMAP* does not attempt to predict absolute pollutant delivery, nor does it attempt to account for temporal variations. Rather it aims to highlight areas where overall topographic and land cover conditions indicate a potential high risk of pollutant delivery. These risks may not always translate into pollution occurrence in reality, e.g. due to the presence of hedgerows or other barriers acting as sediment traps. As such, the monitoring of suspended sediment, phosphates etc, at a limited number of sites, through time, may not be the most appropriate approach to model validation. Instead, the model's performance has been principally assessed according to its ability to explain spatial variance in the distribution of salmonid populations. In other words, is the *SCIMAP* output ecologically significant in terms of salmonids? Some subsequent validation at selected sites has been undertaken using water quality analysis and gravel siltation mapping to aid in confirmation of cause and effect relationships inferred between salmonid populations and landscape controls.

5.3 Results

Figure 5.4 presents the risk maps produced for the Eden catchment, clearly indicating a number of sub-catchments which have a risk of diffuse pollution delivery. Visual comparison of Figure 5.4 (a) (risk of hydrological connectivity) and Figure 5.4 (b) (risk of hydrological connectivity weighted by land cover) suggests that there is a strong correlation between those sub-catchments and reaches of the channel that are highly connected to the landscape and which also have a high proportion of "risky" land cover within their upslope contributing area.

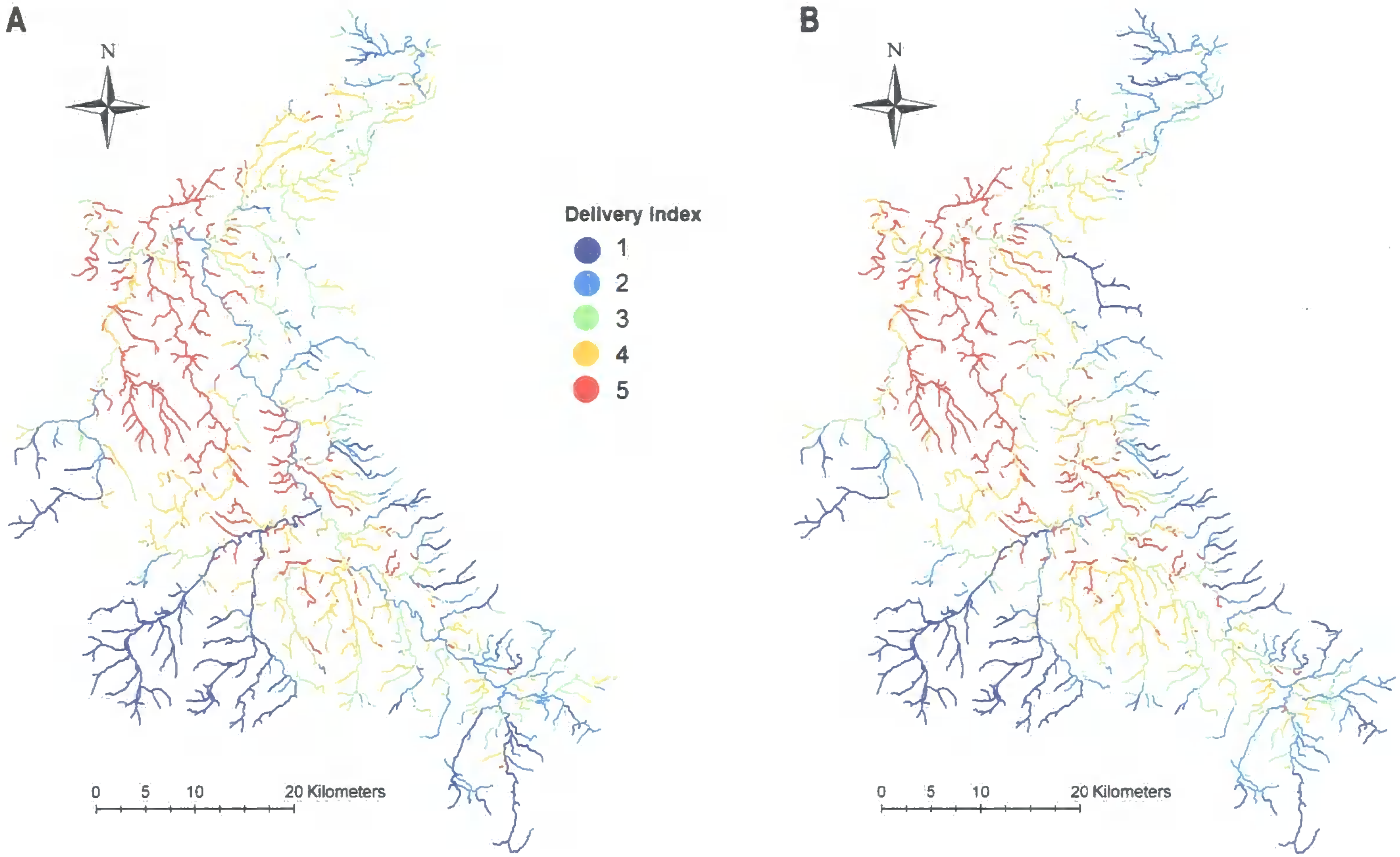


Figure 5.4: *Maps of relative delivery index risk for the River Eden catchment. (a) Risk of surface hydrological connectivity (b) Risk of surface hydrological connectivity weighted by land cover. Risk is classified into five equal membership delivery classes ranging from 1 (locations with the least likelihood of delivery per upslope contributing area) and 5 (locations with the most likelihood of delivery per upslope contributing area).*

Statistical analysis using Spearman Rank correlation confirms that the two factors are correlated (Table 5.2). A non-parametric test was used as the data were found to be non-normally distributed.

Table 5.2: *Relationship between the delivery index for surface hydrological connectivity alone and the delivery index for surface hydrological connectivity weighted by land cover as assessed using Spearman ranked correlation. Tests were two-tailed.*

Year	No. of cases	Correlation co-efficient	*p*-value
2002	351	0.919	0.000
2003	275	0.878	0.000
2004	279	0.920	0.000
2005	291	0.927	0.000
2006	246	0.936	0.000

Although this correlation may partly be because both indices are weighted by the same dilution potential there is also likely to be a relationship between hydrological connectivity and land cover related to the topographic structure of the landscape. Areas predicted as having high surface connectivity are typically found within wide, gently sloping landscapes with relatively long hillslopes where a gentle slope and large upslope contributing area result in water accumulation, saturation and the generation of overland flow. These are also the landscapes most commonly associated with intensive agricultural practices due to easier machine operation and deeper more fertile soils. It could be argued that these areas of the landscape are likely to have been drained reducing the amount of surface flow. However, they are still likely to represent areas of high hydrological connectivity as a result of rapid water flow through drains to the channel network. Research has suggested that tile drains creating shallow sub-surface flow can be a major source and delivery route of contaminants such as fine sediment, and phosphorus attached to sediment (Walling *et al.*, 2002). Areas of steep terrain are less likely to become saturated or be drained as water infiltrating into the soil is more likely to flow away laterally under gravity reducing the likelihood of saturation and overland flow generation. These areas are also more likely to be farmed less intensively. However, a number of subtle differences in the datasets are apparent upon closer inspection. For example, Figure 5.5 demonstrates how risk may change when weighted by land cover. In tributary (a) (red circle) relative risk is reduced compared with tributary (b) (green circle) as despite a relatively high risk of connection the land use is relatively low risk extensive pasture. High levels of connectivity are predicted not to translate into high levels of delivery as a result of lower fine sediment generation within the landscape. In comparison, in tributary (b), relative risk is increased as the presence of relatively high risk intensive pasture means that moderate to low levels of connectivity can still translate into high delivery risk.

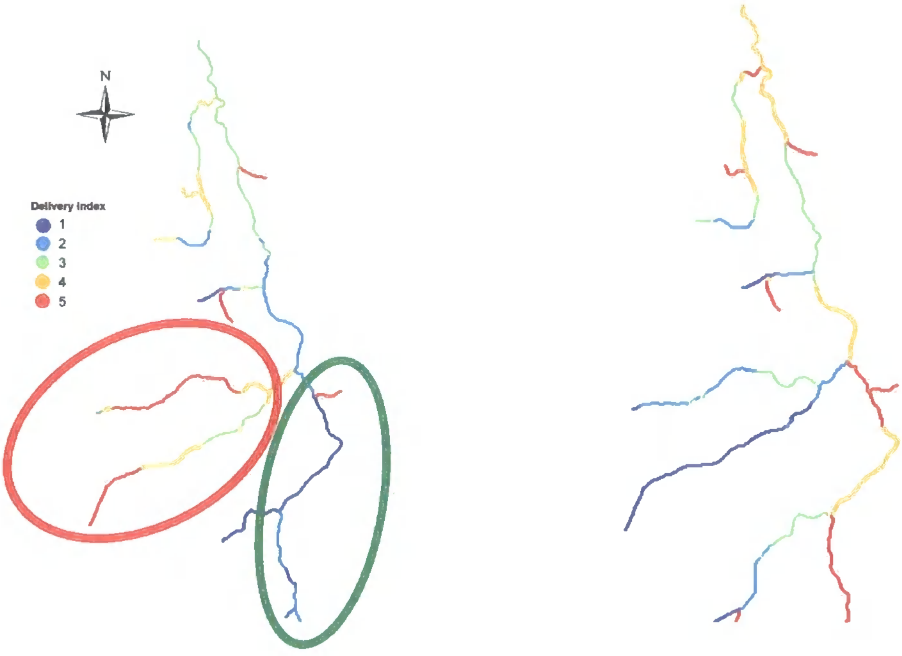


Figure 5.5: *Differences in the delivery index when (a) un-weighted and (b) weighted by land cover*

Throughout the remainder of this chapter the emphasis will be on analysis using the delivery index weighted by land cover.

5.4 Comparison of the *SCIMAP* delivery index with salmonid population data

To assess the role of hydrological connectivity in structuring the relationship between salmonid populations and land cover at the catchment-scale the delivery index weighted by land cover has been related to the spatial distribution of salmonid presence/absence and abundance data. Salmonid population data were provided by the Eden Rivers Trust and Environment Agency as described in Chapter Three (Section 3.3 and Appendix 1). The data used here were salmon and trout fry presence/absence and abundance (number caught in 5 minutes) for 2002-2006. It should be noted that in 2006, the electrofishing survey focused on trout and as such concentrated on surveying the smallest tributaries within the Eden catchment. The 2006 data are therefore unlikely to be suitable for assessing salmon populations which are more likely to be found within larger tributaries and the main river. The data were provided as a series of ArcGIS point shapefiles, one for each year. Each file was snapped to the output delivery index layers produced by the *SCIMAP* model. The delivery index at each electrofishing site was then extracted to the point shapefile using the "extract to point tool" included in the ArcGIS toolbox. The resulting attribute table was then exported to SPSS v.12 for analysis.

Initial analysis was undertaken to see whether the delivery index weighted by land cover could discriminate between sites where salmonid fry were present and those where salmonid fry were absent (Table 5.3). Results are reported for scenario two (risk of hydrological connectivity weighted by land cover). The mean delivery index was recorded for sites with and without fry present, and Mann Whitney tests were used to examine whether there was a statistically significant difference in the central tendency of the delivery index. Two-Sample Kolmogorov-Smirnov tests were also used to test whether there was a difference in the distribution (location and shape) of the delivery index between sites with and without salmonid fry. Non-parametric tests were selected due to the non-normal distribution of the data which could not be corrected for using standard transformation procedures.

Table 5.3: *Comparison of the delivery index statistics for sites with fry present and sites with fry absent stratified by species.*

Year	Species	Mean delivery index weighted by land cover		*p* value	
		Fry present	Fry absent	Mann Whitney	Kolmogorov-Smirnov
2002	Trout	0.0450 ± 0.0024, n = 167	0.0587 ± 0.0025, n = 184	<0.0001	0.0003
	Salmon	0.0506 ± 0.0021, n = 204	0.0543 ± 0.0031, n = 147	0.937	0.050
2003	Trout	0.0521 ± 0.0022, n = 192	0.0762 ± 0.0046, n = 83	<0.0001	<0.0001
	Salmon	0.0575 ± 0.0024, n = 154	0.0619 ± 0.0040, n = 121	0.773	0.018
2004	Trout	0.0501 ± 0.0024, n = 161	0.0627 ± 0.0033, n = 118	0.003	0.014
	Salmon	0.0552 ± 0.0022, n = 139	0.0557 ± 0.0033, n = 140	0.233	0.002
2005	Trout	0.0521 ± 0.0027, n = 140	0.0599 ± 0.0030, n = 151	0.089	0.105
	Salmon	0.0561 ± 0.0026, n = 168	0.0562 ± 0.0034, n = 123	0.763	0.793
2006	Trout	0.0564 ± 0.0033, n = 141	0.1086 ± 0.0060, n = 105	<0.0001	<0.0001

Table 5.3 shows that the delivery index weighted by land cover discriminates extremely effectively between the presence and absence of trout fry most notably in 2002, 2003, 2004 and 2006. The mean delivery index is consistently lower for sites where trout fry are present compared with those where trout fry are absent. Mann Whitney and Kolmogorov-Smirnov tests reveal that these differences are statistically significant ($p<0.05$) for all years except 2005. On first inspection the delivery index does not seem so effective at discriminating between sites where salmon fry are present and absent. Differences in the mean delivery index are slight and inconsistent across years, whilst Mann Whitney tests are not significant ($p>0.05$) in any year. However, on closer examination the Kolmogorov-Smirnov tests for sites with and without salmon fry do report a statistically significant difference in the distribution (shape and location) of the

delivery index for all years excepting 2005. This suggests that there may still be a relationship between the delivery index and salmon fry but that it is non-linear.

To investigate the ecological significance of the delivery index further, salmonid abundance data were also examined. Following Lane *et al.* (in draft), the delivery index weighted by land cover was classified into five equal membership delivery classes ranging from 1 (locations with the least likelihood of delivery per upslope contributing area) and 5 (locations with the most likelihood of delivery per upslope contributing area). Graphs of the cumulative frequency distribution of fry abundance for the classified delivery index were then produced for each year (Figures 5.6 and 5.7). The results proved very promising, and highlighted that there was indeed a striking relationship between the delivery index weighted by land cover and fry abundance. However, one of the most interesting and unexpected findings was that this relationship differed according to species. Trout fry demonstrated a negative linear relationship with the delivery index. In general, sample points in the lowest delivery index class (1) had a higher level of trout fry abundance than those in the higher delivery index classes (2-5). This distinction was clearest in 2002, 2005 and 2006. In 2003 and 2004, the lowest two delivery index classes (1 and 2) were less distinct from one another but still had higher trout fry abundance than the higher delivery index classes (3-5). The highest delivery index class (5) consistently had lower trout abundance than the lowest delivery index class (1) across all years. In contrast, salmon fry, demonstrated a non-linear relationship, in that both low and high levels of the delivery index corresponded with low abundance of salmon fry, with an optimum level for performance in the central range. In general, sample points in the lowest and highest delivery index classes (1 and 5) had lower salmon fry abundance than those in delivery index classes (2-4). This distinction was clearest in 2002, 2003 and 2004. In 2005 sites within the highest delivery index class (5) had higher abundance similar to that in classes 2-4, whilst sites in the lowest delivery index class (1) had the lowest salmon fry abundance of all.

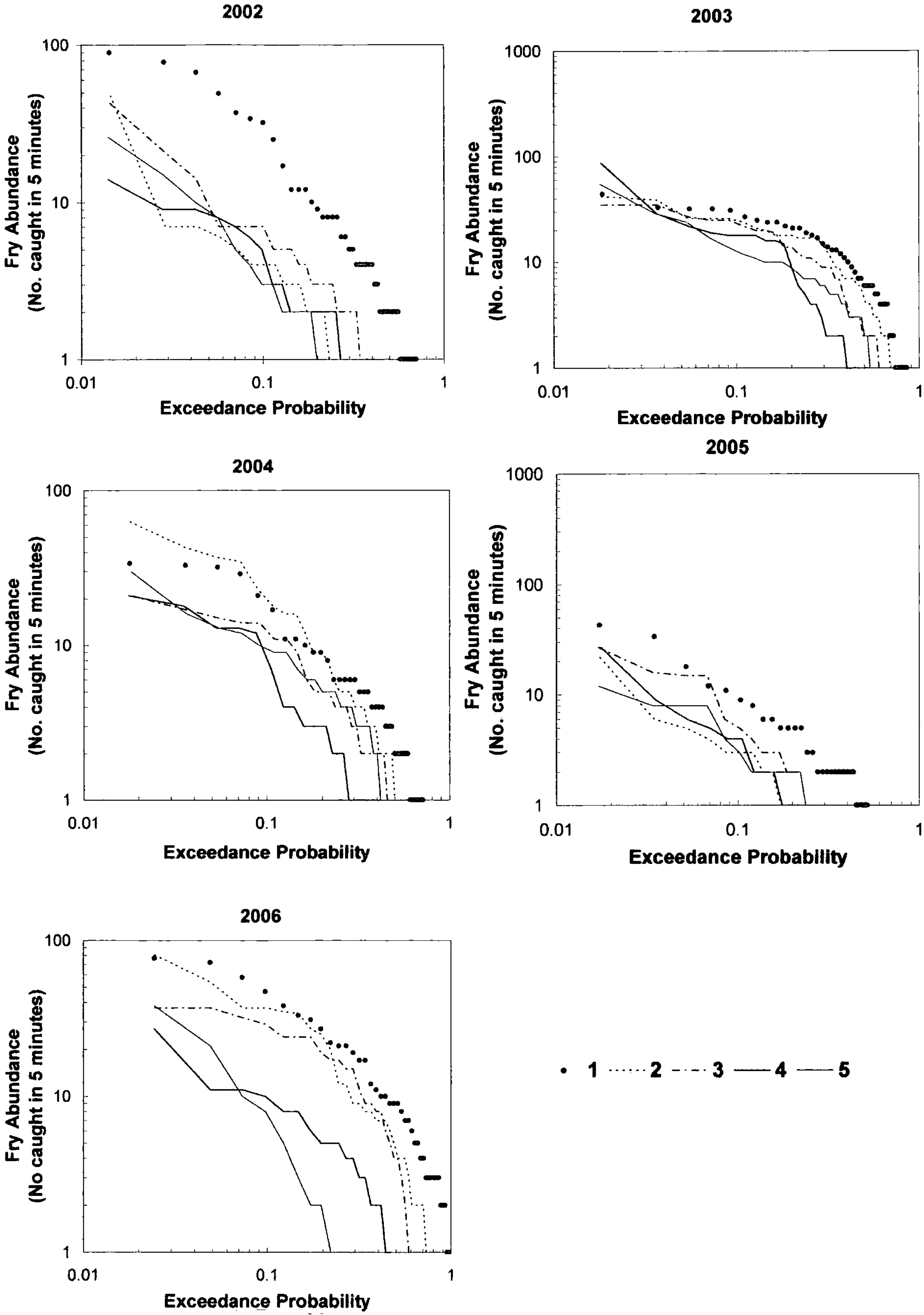


Figure 5.6: *Cumulative frequency distributions of trout fry abundance for the classified delivery index: 1 indicates relatively low predicted levels of delivery from land units upstream increasing to 5 which indicates relatively high predicted levels of delivery. (After, Lane et al., in draft)*

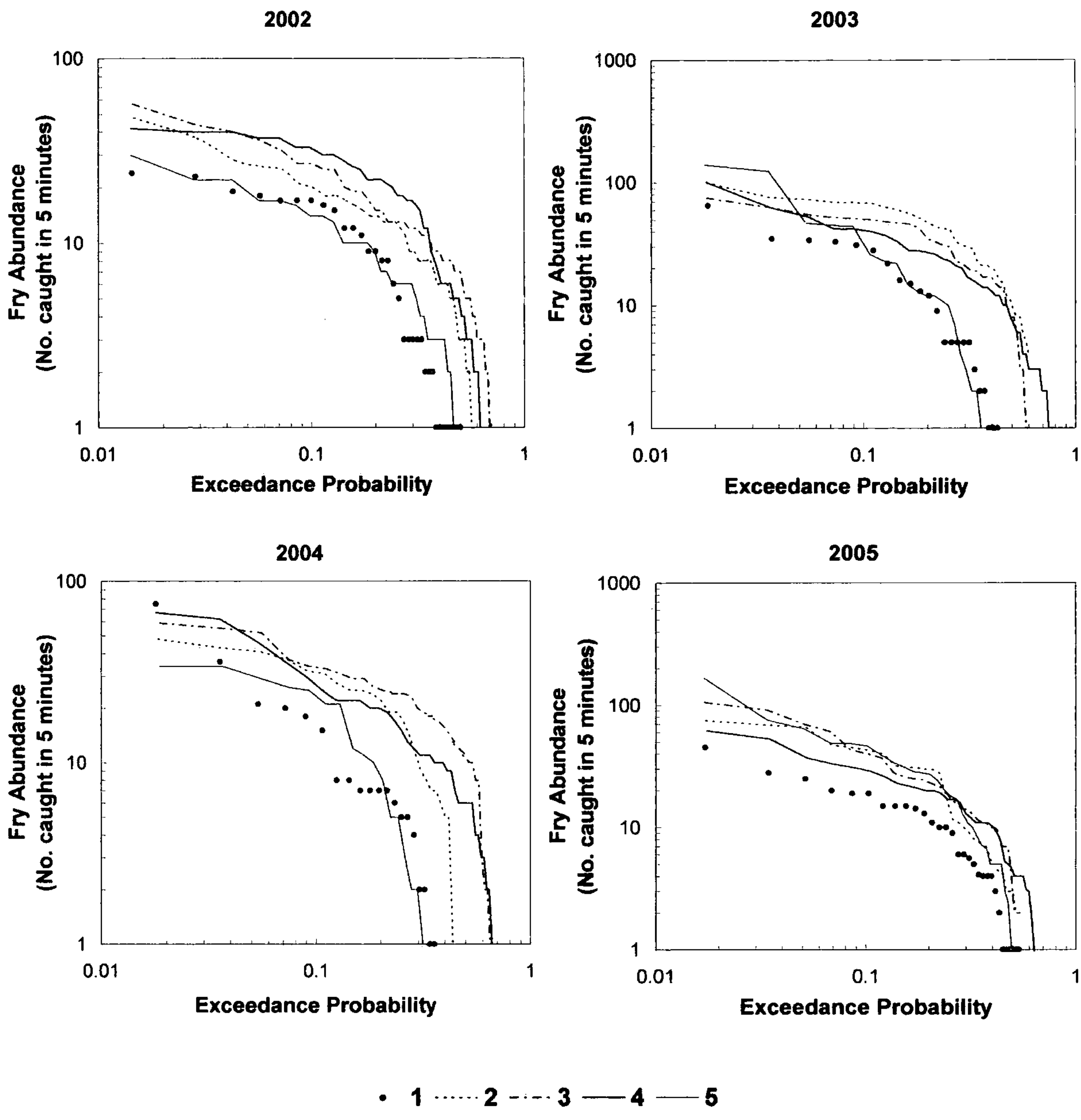


Figure 5.7: *Cumulative frequency distributions of salmon fry abundance for the classified delivery index: 1 indicates relatively low predicted levels of delivery from land units upstream increasing to 5 which indicates relatively high predicted levels of delivery. (After, Lane et al., in draft)*

What appears clear from these results is that the degree of hydrological connectivity between "risky" land covers within the catchment and the channel network, as controlled by surface topography, is important in structuring salmonid populations at a catchment-scale. As the salmonid data are based upon semi-quantitative sampling they are likely to contain a considerable degree of spatial noise and it is particularly surprising that such clear presence/absence and abundance signals were observed. Two potential mechanisms that may be inferred to explain why this landscape variable is so important relate to the impact of upslope land cover as a proxy for agricultural activity upon: (1) soil erosion and fine sediment delivery to the channel; and (2) nutrient production and delivery to the channel. With regard to (1), soil erosion and fine sediment delivery have been associated with the poor survival to emergence of salmonids due to an increased infiltration of fines into redds located in areas of high fine sediment loading (e.g. Crisp, 1996; Summers *et al.*, 1996; Soulsby *et al.*, 2001; Greig *et al.*, 2005, see Chapter 2 Section 2.5.1). As fry typically remain within a few hundred metres of their spawning site it is likely that lower abundance of salmonid fry will be observed in areas of high fine sediment delivery compared with areas of low delivery. Research has shown that catchment sources (diffuse pollution) can contribute a significant proportion to in-stream fine sediment loadings and that the dominant transport pathways to the channel are surface and shallow sub-surface flows (Walling *et al.*, 2002). Within the *SCIMAP* model, land cover is ranked according to its perceived risk of soil erosion, whilst the explicit treatment of surface hydrological connectivity considers the likelihood of delivery and filters the relationship between the land surface and channel network. As such, it is hypothesised that reaches with a high delivery index weighted by land cover and low salmonid abundance will also demonstrate a high loading of fine sediment. Whilst this mechanism may help to explain low numbers of salmon and trout fry at sites with a high delivery index, it does not explain the different response observed between salmon and trout fry at low values of the delivery index.

The second mechanism inferred here is that the delivery index is a proxy for the delivery of nutrients to the channel. Similar to fine sediment, research has shown that catchment sources can contribute a significant proportion to in-stream nutrient concentrations where point sources have been addressed, and that in the case of certain nutrients, most notably phosphorus, that surface and shallow sub-surface flow pathways can be an important delivery route (Heathwaite *et al.*, 2005). It is likely that the *SCIMAP* classification of land cover according to soil erosion risk may also be representative of the risk of excessive nutrient loading to the land. Arable and intensively managed grassland received the highest risk weighting. These land covers are also likely to receive the highest nutrient loadings as a result of fertiliser, manure and slurry application when compared with extensively managed pasture, heath and moorland which were ranked at low risk. Reaches with a high delivery index will have a higher concentration of nutrients than those with a low delivery index. Salmonids require water with a high dissolved oxygen content at all stages in their life-cycle and excessive nutrient concentrations, leading to eutrophication and increased biological oxygen demand have been suggested as a significant hypothesis for declining populations (Heaney *et al.*, 2001). Again, this mechanism may help to explain low numbers of salmon and trout fry at sites with a high delivery index, but note the difference in species response in the lowest delivery index class. It is suggested that this may be explained by a variation in feeding habits between the two species (Section 2.5.2.3) in relation to nutrient delivery. Salmon are particularly dependent on invertebrates supported by autochthonous production (O'Grady, 1993). Low nutrient input will result in reduced autochthonous production, which at the lowest delivery index levels may constrain salmon fry abundance. On the other hand, trout are more opportunistic and will also feed on terrestrial invertebrates supported by allochthonous production where in-stream production is limited. As such, their abundance may be less restricted in reaches of low delivery index. Within the Eden catchment the lowest delivery index values are predicted predominantly within the Ullswater and Lowther Valley area, where resistant volcanic geology further accentuates oligotrophic conditions resulting in a particularly marginal environment for salmon.

The Ullswater and Lowther Valley area has been historically renowned for supporting populations of multi-sea-winter (MSW) spring-run salmon as confirmed by radio-tracking, but it is generally perceived that these are in decline (Gowans, 2004). This decline is not unique to the River Eden but has also been observed elsewhere (e.g. Consuega *et al.*, 2005; Quinn *et al.*, 2006). Factors such as climatic variations at sea (Martin and Mitchell, 1995; Boyle and Adams, 2006) and the selective exploitation of spring-run fish (Consuegra *et al.*, 2005; Quinn *et al.*, 2006) have been suggested as explanations for the changes observed in the length of time spent at sea and migratory run times (Section 2.3). Three other possible reasons for the decline in multi-sea-winter spring-run fish returning to the Ullswater/Lowther Valley area, associated with the freshwater environment are suggested here. First, the valley floor in this area has a long history of being heavily controlled (Orr and Newson, 2003) with localised canalisation and flood protection. This may have resulted in a loss of in-stream habitat diversity for both juveniles and returning adults. Early arriving salmon spend longer in the freshwater environment and may therefore experience greater stress as a result of habitat degradation, particularly if this is associated with the upland environments in which they typically spawn (Quinn *et al.*, 2006). Second, canalisation may have exacerbated the impact of low nutrient delivery on juvenile salmon abundance by disconnecting the channel from its floodplain reducing nutrient exchange. However, chemical monitoring in Ullswater has found nutrient concentrations to be rising in the lake not falling which would suggest that this is not be case (Zinger-Gize *et al.*, 1999). Third, and alternatively, it has been suggested that juvenile growth rates may be linked to the tendency for female salmon to return to the river as grilse or multi-sea-winter fish, with lower growth rates corresponding to a longer period at sea (Shearer, 1993). In marginal environments, such as the Ullswater area, lower juvenile growth rates, caused by lower food availability, may have resulted in a greater tendency for fish to remain at sea for more than one winter. Visual comparison of the spawning location of radio-tagged fish, in 1999 and 2000, stratified by sea-age (Section 3.3.4) and the delivery index weighted by land cover indicates that there is a strong correlation between the two factors (Figure 5.8). This is confirmed by statistical testing (Table 5.4) which shows a significant negative correlation between the delivery index weighted by land cover and sea-age. In other words, as salmon are understood to typically return to their river of origin (natal stream) to spawn (Armstrong *et al.*, 2003), areas with a low risk of nutrient delivery appear to have a higher probability of producing multi-sea-winter fish.

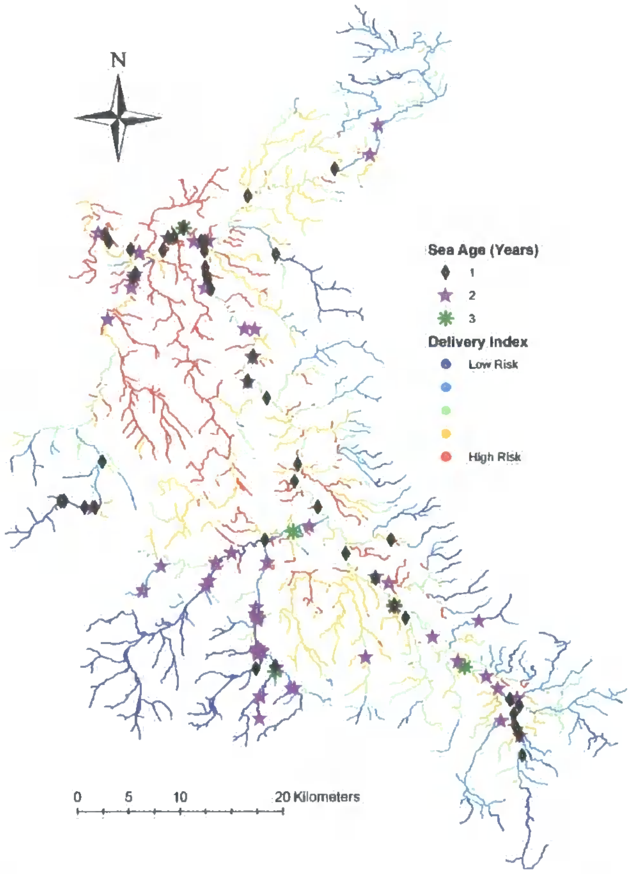


Figure 5.8: *Spawning location stratified by sea-age mapped onto the delivery index weighted by land cover*

Table 5.4: *Statistical testing of the relationship between the delivery index weighted by land cover and spawning location stratified by sea-age. (Sample size = 125).*

	Analysed by number of years at sea (1 year, 2 year or 3 year)		Analysed by age category (Grilse or Multi-Sea-Winter)	
	Spearman rank correlation	Kruskal-Wallis Test	Mann-Whitney Test	Two-Sample Kolmogorov-Smirnov
Co-efficient	-0.259	8.379	1353.5	1.715
p-value	0.004	0.015	0.004	0.006

It is suggested that, if nutrient and hence food availability in such areas increases due to pressures from agricultural intensification and/or rural expansion, that juvenile growth rates may also increase. If juvenile growth rates are linked to the time spent at sea then an increase in nutrients could potentially result in an increased propensity for salmon to return to the river as grilse. Grilse are likely to produce fewer and smaller eggs than MSW fish (Quninn *et al.*, 2006) and due to their later entry into the river system (summer/autumn as opposed to spring) they may also be less likely to migrate the considerable distance required to return to this area of the Eden catchment (Gowans, 2004). If this is the case, a gradual decline in egg deposition in the Ullswater area may be resulting in a decline in the population. These characteristics are primarily considered to be genetically determined, but if environmental influences in food availability are also important, then changing nutrient and food availability may have considerable consequences for fisheries managers particularly in sensitive marginal environments (Armstrong *et al.*, 1998). As noted in Chapter Two (Section 2.5.2.3), there is currently a lack of knowledge as to whether changes to food availability result in changing growth rates, changing abundance or both, and whether changing growth rates influence migratory behaviour. These are only potential explanations for the declining numbers of salmon observed in the Ullswater area and additional research would be required to evaluate the situation further. However, in an area that is naturally marginal in terms of supporting salmon populations any one of these changes (habitat, climatic or exploitation) may be significant enough to cause a decline in populations.

Both trout and salmon fry failed to show a strong relationship with the delivery index in 2005, and it is suggested that this may be the result of the extreme January 2005 floods which took place within the Eden catchment. Floods which occur after salmonids have spawned can destroy developing eggs and reduce recruitment (Jowett and Richardson, 1989) through wash out of redds. If redd damage occurred in the Eden catchment during the January 2005 floods, it could have obscured the relationship between the delivery index and fry abundance. Research by Brown (2006a) found numbers of trout fry to be significantly reduced in 2005 compared with the previous three years, whilst salmon numbers remained unchanged, suggesting that egg (specifically trout egg) damage may have occurred. The degree to which egg damage occurs is related to the depth at which eggs are laid. Brown trout have an egg burial range of 8-25 cm and are therefore more susceptible to redd washout than salmon which have a burial range of 13-30cm (Crisp 1989). An alternative explanation, which may account for the increase in salmon abundance within the highest delivery index class in 2005, is that the flood may have "cleaned" the gravel substrate, removing accumulated silt and reducing the negative impact upon salmonids.

The results presented here support the theory that catchment-scale processes including land cover, as filtered by the topographical structuring of hydrological response, are important in terms of explaining the spatial pattern of salmonid abundance. They also suggest that the *SCIMAP* model is an effective and ecologically relevant tool for the assessment of landscape risk within catchments. To evaluate model performance further, gravel siltation mapping and water chemistry analysis were undertaken at selected sites to assess whether supporting evidence could be found for the two mechanisms inferred to explain the importance of land cover and hydrological connectivity. Again focus is on the delivery index weighted by land cover.

5.5 Gravel siltation mapping

One of the primary mechanisms inferred to explain the relationship observed between the delivery index and salmonid performance is the effect of upstream agricultural activity on soil erosion and fine sediment delivery to the channel. To test this hypothesis gravel siltation mapping within spawning and juvenile salmonid habitat has been related to the delivery index at a catchment scale. Data on gravel siltation were obtained from the Eden Rivers Trust which recorded the data during electrofishing surveys undertaken in 2004, 2005 and 2006. The presence of siltation within surveyed riffles was assessed both visually by inspection of substrate appearance, and mechanically by the surveyor digging their heal into the substrate to test its cohesion. In areas impacted by the excessive infiltration of fines, the channel bed is likely to become concreted preventing coarse substrate from being easily dislodged during this mechanical action. The technique was highly subjective but it did enable data on gravel siltation to be collected rapidly and cheaply across a large number of sites (166-279 sites being assessed in each year). All assessments were made by the same person helping to reduce subjective error. The data were plotted within ArcGIS as a point shape file and snapped to the *SCIMAP* risk layers. The delivery index for scenario two (risk of hydrological connectivity weighted by land cover) at each sampling point was then extracted to the point shapefile using the "extract to point tool" included in the ArcGIS toolbox. The resulting attribute table was then exported to SPSS v.12 for analysis.

5.5.1 Gravel siltation mapping results

Encouragingly, the results of the gravel siltation analysis support the hypothesis that reaches with a predicted high delivery index have a greater probability of fine deposition and siltation of gravels than those with a low predicted delivery index. Figure 5.9 (a) reveals that, for all three years, the median delivery index was lower for sites with no gravel siltation when compared with sites where gravel siltation was observed. Mann Whitney tests applied separately to each year confirmed that there was a statistically significant difference between the delivery index for sites with and without siltation (Table 5.5). Again a non-parametric test was selected due to the non-normal distribution of the data, which could not be corrected using standard transformation procedures.

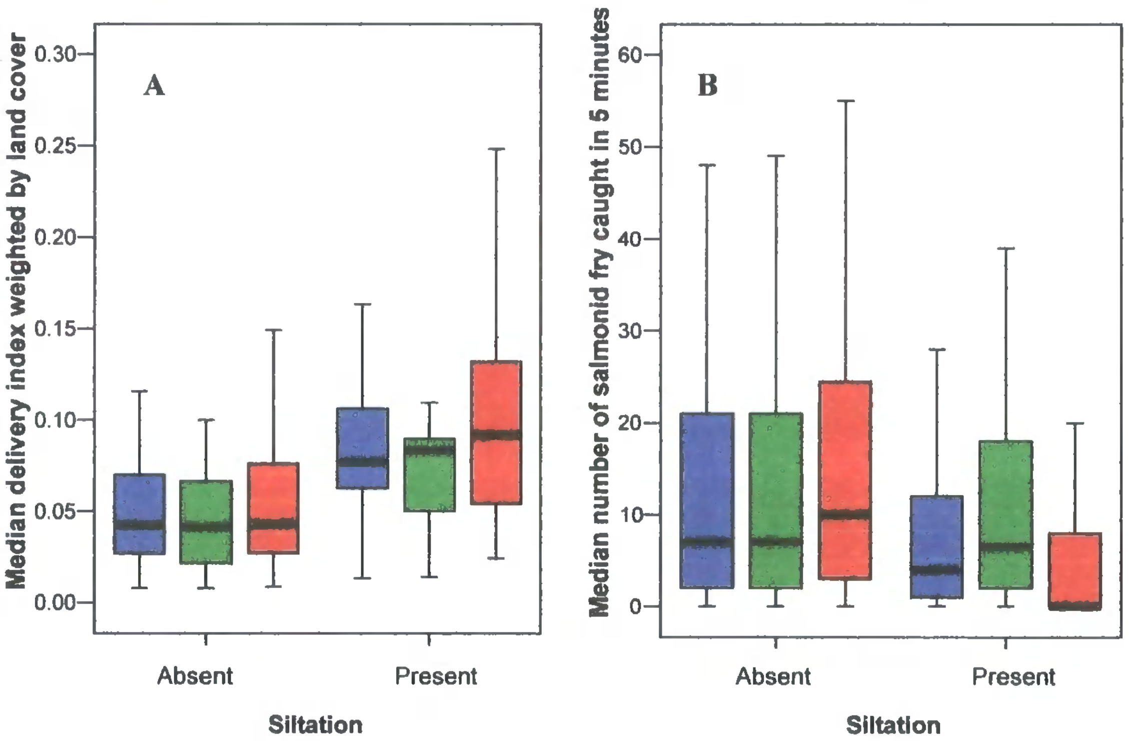


Figure 5.9: *Boxplots showing the median, inter-quartile range and distribution of (a) the delivery index weighted by land cover and (b) salmonid fry abundance for sites with and without gravel siltation.*

Salmonid fry abundance was also found to differ between those sites with siltation and those without. Sites with observed gravel siltation consistently reported lower numbers of salmonid fry than those without siltation (Figure 5.9(b)). This discrimination was found to be statistically significant (Mann Whitney two-tailed tests) in both 2006 (99.9% confidence level) and 2004 (98% confidence level), but not in 2005. Again, this lack of association in 2005 may reflect the impact of the January 2005 extreme flood which may mask or reduce the impact of habitat parameters such as gravel siltation.

Table 5.5: *Mean and standard error of delivery index values and salmonid abundance for sites with and without gravel siltation, together with the results of Mann Whitney statistical testing.*

Year	Parameter	Siltation Absent	Siltation Present	*p* value
	Delivery index‡	0.049 ± 0.0019	0.0843 ± 0.0054	<0.0001
2004	Salmonid abundance†	13.19 ± 1.044	8.16 ± 1.691	0.017
	Sample size	230	49	
	Delivery index	0.047 ± 0.0026	0.073 ± 0.0067	<0.0001
2005	Salmonid abundance	15.64 ± 1.77	11.87 ± 2.94	0.694
	Sample size	148	18	
	Delivery index	0.054 ± 0.0036	0.104 ± 0.0053	<0.0001
2006	Salmonid abundance	17.85 ± 2.06	6.18 ± 1.11	<0.0001
	Sample size	123	122	

‡ Delivery index for scenario two, risk of surface hydrological connectivity weighted by land cover.

† Number of salmonid fry caught in 5 minutes of semi-quantitative electrofishing

These results suggest that the level of gravel siltation is important in determining salmonid performance within the Eden catchment. Further, they suggest that the level of gravel siltation within individual reaches may be related back to landscape parameters such as the type of upstream agricultural activity (which was classified according to its perceived risk of soil erosion) and the degree to which reaches are hydrologically connected to these activities. If this is the case then management activities which aim to reduce the risk of erosion (e.g. soil conservation strategies) or to reduce the amount of fine sediment delivered (e.g. riparian buffer zones) within hydrologically connected areas of the catchment may have a beneficial impact on in-stream habitat and salmonid performance. This emphasises the importance of considering the role of landscape-scale processes when evaluating local in-stream habitat conditions and ecological performance. These results also add support to the use of the *SCIMAP* model as an appropriate and ecologically relevant tool for determining the impact of landscape scale controls upon in-stream habitat and ecological productivity, and for the prioritisation of land management activities at a catchment-scale.

5.6 Water chemistry

The second mechanism inferred to explain the relationship between the delivery index and salmonid performance was the influence of upstream agricultural activity and surface/shallow sub-surface hydrological connectivity upon the level of nutrients delivered to the channel network. To assess this, an intensive spatial sample of water quality was collected for part of the upper

Eden catchment. 210 water samples were collected by 15 samplers on 7 September 2005 between 9:00 and 13:00. The samples were collected during low flow conditions following a period of sustained dry weather. Figure 5.10 illustrates the distribution of sampling points which were spread across the range of delivery index values determined. Unfortunately, fewer samples than intended were collected from reaches with the highest delivery index as these were frequently found to be dry. The majority of samples were focused around confluences (e.g. one ~ 50m upstream, one ~50m downstream and one in the inflowing tributary) to allow analysis of changing risk and water quality as inputs of differing risk join the channel network. Samples were collected in 50ml vials and Appendix Three records the standard sampling protocol adopted by all samplers. Samples were transported with freezer packs to Durham University on the day of collection and refrigerated within 8 hours of sampling. Prior to analysis the samples were filtered using a 0.2 micron syringe filter. They were subsequently analysed on a Dionex DX500 ion chromatography system using suppressed conductivity as the detection system. Data were recorded for a range of cations and anions including phosphate, potassium, nitrate and ammonium.

Delivery of material by surface and shallow sub-surface lateral flow is typically associated with rainfall and high discharge events. However, changes in water quality during such periods can be rapid and it was felt that the spatial integrity of the samples would be more realistically captured through low flow sampling, as it is relative risk not absolute risk that is being examined. Whilst peaks in contaminant concentration may not be captured the assumption was made that low flow samples could still reflect spatial variation in the delivery of nutrients as certain contaminants will persist within the environment over time. Water quality results were plotted as a point shapefile in ArcGIS with all points snapped to the *SCIMAP* output layer. The delivery index at each sampling point was then extracted to the point shapefile using the "extract to point tool" included in the ArcGIS toolbox. The resulting attribute table was then exported to SPSS v.12 for analysis.

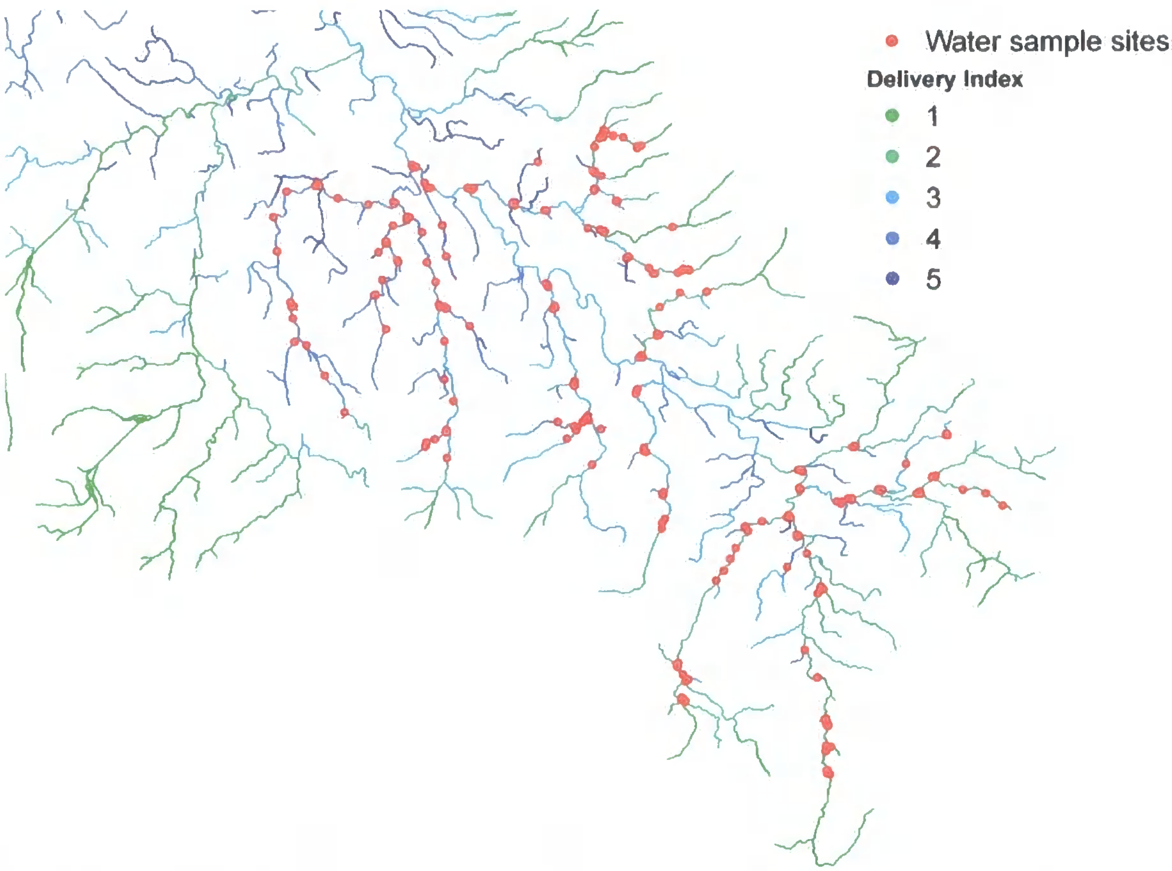


Figure 5.10: *Location of water sampling sites overlying the delivery index weighted by land cover classified into 5 equal membership classes ranging from 1 (locations with the least likelihood of delivery per upslope contributing area) and 5 (locations with the most likelihood of delivery per upslope contributing area).*

5.6.1 Water chemistry results

Initial visual inspection of the results using scatterplots indicated that there was a relationship between the nutrient concentrations measured and the delivery index representing the risk of surface hydrological connection weighted by land cover (Figure 5.11). This was confirmed by statistical testing using Spearman Rank correlations (Table 5.6). As above, a non-parametric test was selected due to the non-normal distribution of the data.

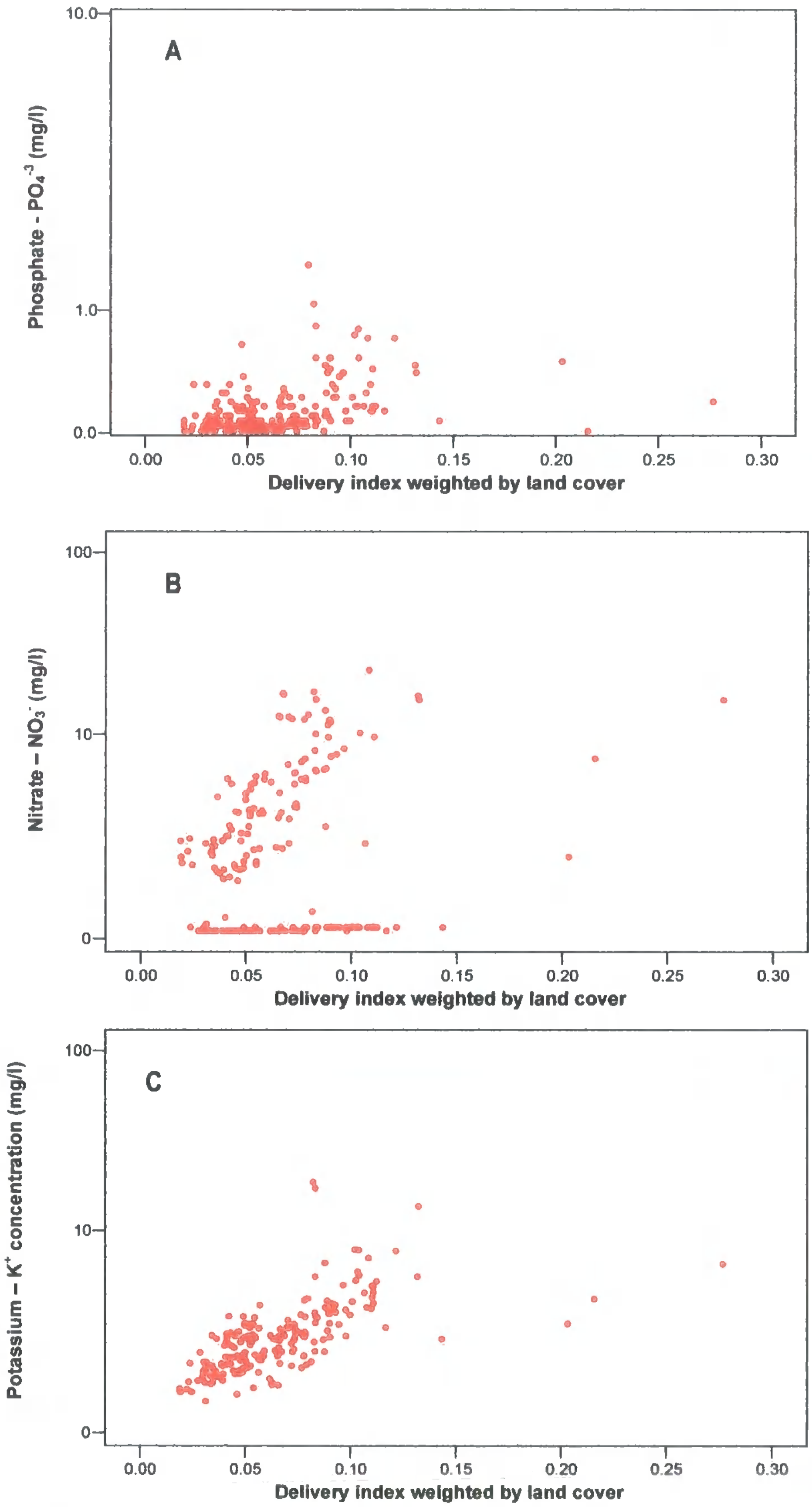


Figure 5.11: *Scatterplots of nutrient concentration (mg/l) against delivery index weighted by land cover within the Eden catchment (a) phosphate (PO_4^{-3}), (b) nitrate (NO_3^-) and (c) potassium (K^+).*

Table 5.6: *Spearman Rank correlation results between the SCIMAP delivery index weighted by land cover and concentrations of potassium, phosphate, and nitrate. Tests were two-tailed.*

Delivery Index	Potassium	Phosphate	Nitrate
Correlation coefficient	0.757	0.453	0.301
p value	<0.0001	<0.0001	<0.0001

The correlation coefficient in all cases was positive confirming the assumption that increased surface hydrological connection to "risky" land covers increases the probability of higher nutrient delivery and concentrations in receiving waters. Of the three ions measured, potassium concentrations showed the strongest correlation with the delivery index. Potassium is commonly found to excess within agricultural soils as a result of over-application, especially upon arable or intensively managed grassland units where combined NPK fertilisers are used. Application rates are commonly calculated to meet nitrogen and phosphorus demands resulting in surplus loadings of potassium. Supply is therefore unlikely to limit delivery as potassium is generally available and easily mobilised wherever there is shallow sub-surface runoff from such units. It is also a relatively stable ion which is likely to persist within the environment over time being detectable in low flow water samples. For these reasons potassium concentrations were used to examine trends in the delivery index weighted by land cover and nutrient concentrations downstream. As illustrated by Figure 5.12 downstream variations in potassium concentration generally reflect downstream variations in the delivery index for both the main stem River Eden and Trout Beck. Inputs from tributaries that are predicted to increase the delivery index in the main channels generally correspond with an increase in potassium concentration. To a lesser degree, inputs that decrease the predicted delivery index also correspond with decreases in potassium concentration. Also evident is an increase in the variability of potassium concentrations at higher values of the delivery index. This may reflect an increase in sensitivity to land management (independent of land cover) at high values of the delivery index. At low levels of the delivery index, topographic structuring of hydrological pathways may be the main factor limiting delivery to the channel. However, as the hydrological connectivity increases and no longer limits delivery, land management (independent of land cover) may become more important in controlling actual delivery. For example, two different fields of intensively managed pasture with the same degree of hydrological connectivity may result in different levels of delivery due to differences in management such as stocking levels; manure, slurry and fertiliser spreading practises; the presence of buffer zones; gate position; and tramline orientation (Caruso, 2001). Further research would be required to ascertain that this is the case but it does raise the potential importance of land management as different to land cover or land use as a control on in-stream conditions.

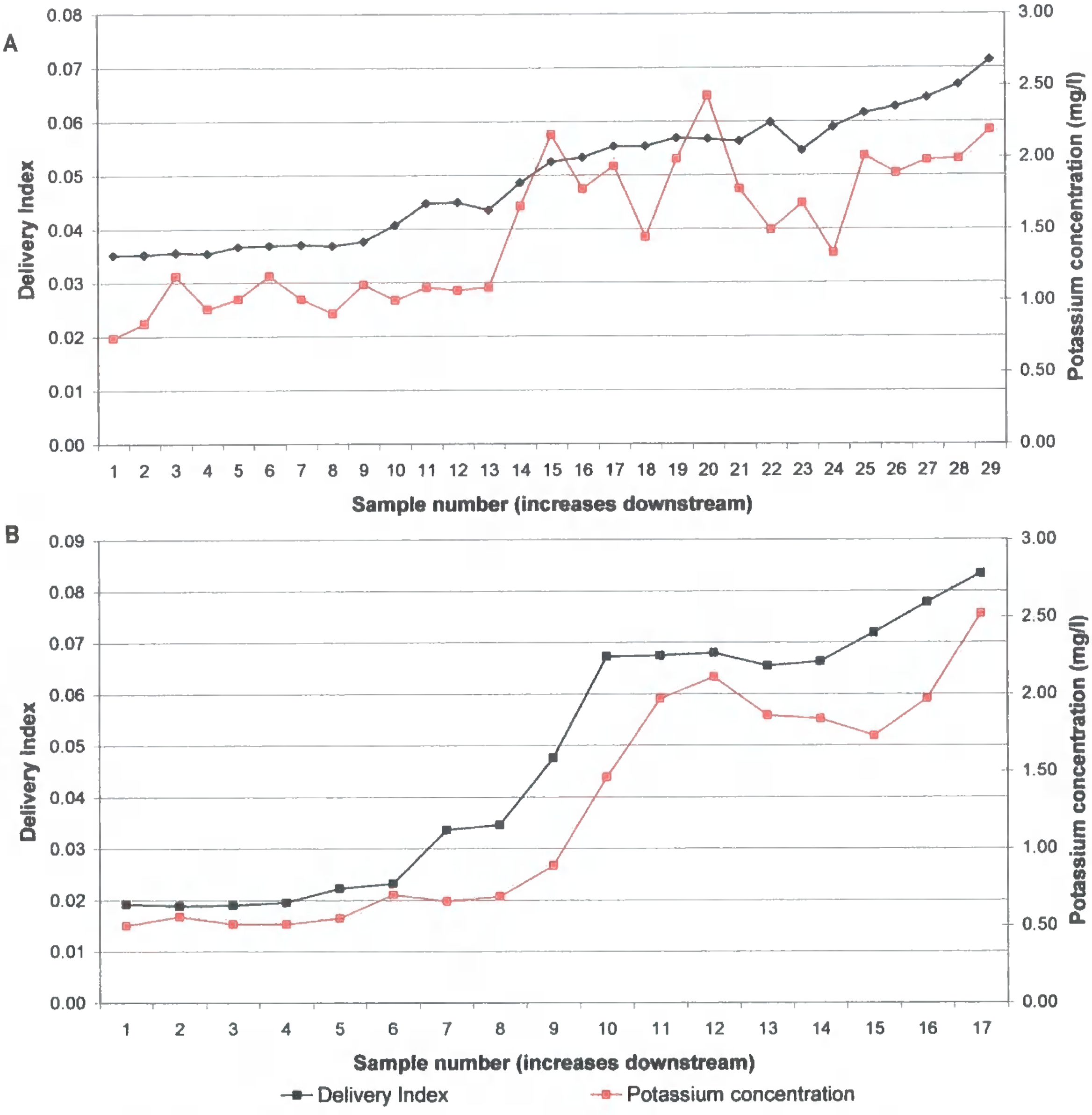


Figure 5.12: *Downstream tends in the delivery index weighted by land cover and potassium concentration. (a) The River Eden and (b) Trout Beck*

Research has shown soil erosion and sediment associated transport by surface and shallow sub-surface pathways to be an important delivery mechanism for phosphates (Russell *et al.*, 1998). This is supported for the Eden catchment by the correlation that was observed between the delivery index and in-stream phosphate concentrations. Less expected was the clear observation of a correlation between nitrate and the delivery index. The delivery index relates to surface hydrological connectivity and delivery of contaminants via overland flow or shallow sub-surface pathways, whereas nitrate delivery is commonly associated with soil leaching processes and deeper sub-surface or groundwater delivery pathways (e.g. Jarvie *et al.*, 2003). Conversely, these results suggest that surface and shallow sub-surface delivery pathways may also be important for nitrate delivery. However, this correlation may also arise because both variables correlate with another causal variable, mostly likely land cover. Both nitrates and phosphates have been associated with eutrophication of freshwaters, which can result in algal blooms, an increase in biological oxygen demand and consequently a reduction in the levels of oxygen available to support salmonids. Ammonia concentrations were also measured, but few samples reported detectable levels (>0.01 mgl^{-1}) and no relationship was observed with the delivery index. A *p* value of 0.349 was reported for the Spearman Rank correlation of ammonia against the delivery index weighted by land cover. Upon entering watercourses ammonia is rapidly broken down to nitrite and nitrate and is unlikely to persist in the environment as ammonia at times of low flow. However, research by Eden Rivers Trust using continuous water quality samplers on the River Lyvennet, predicted to have a relatively high delivery index (risk category 3-4) throughout, has reported peaks in ammonia concentrations corresponding with rainfall events (Brown, 2006c). Although the chemical data are limited, to a small spatial scale this could represent a link with runoff from land where slurry and manures have been recently applied prior to rainfall, both of which are typically high in ammonia. Should rainfall occur before the slurry or manure is incorporated within the soil matrix then it and contaminants contained within it may be transported overland by surface pathways. Consideration of organic nutrient applications may therefore be just as important as inorganic applications when evaluating the risk of diffuse pollution from agriculture. Further investigation would be required to confirm that there is definitely a relationship between the delivery index and the delivery of ammonia.

It appears apparent from these results that the topographical structuring of hydrological response is important in determining the delivery of a range of solutes from the land surface to the channel network. This adds support to the hypothesis that the mechanism linking salmonid performance to the delivery index may be nutrient delivery. However, it should be recognised that the *SCIMAP* approach is unable to discriminate between the risk of fine sediment delivery, the risk of nutrient delivery and the risk of other parameters not considered here. Again results support the use of the *SCIMAP* model as an appropriate and ecologically relevant tool for prioritising the risk from landscape scale controls to in-stream habitat and ecological productivity, and highlight the benefits of considering hydrological connectivity between land parcels and the river system.

5.7 Discussion

The results presented here highlight the importance of considering landscape controls over salmonid habitat. Both the spatial pattern of salmonid presence/absence and abundance within the Eden catchment appear to be structured by catchment land cover as filtered by surface and shallow sub-surface hydrological connectivity. The clarity of the relationships observed was particularly striking considering the spatial noise that is likely to be present within the salmonid data. All the results presented above are based upon the risk of hydrological connectivity weighted by land cover. However, all the statistical tests reported here were also undertaken for the risk of hydrological connectivity alone assuming uniform land cover. Similar relationships were observed stressing the significance of topographically structured hydrological connectivity (e.g. Tables 5.7 and 5.8) in filtering the relationship between land cover and in-stream ecology. However, adding land cover to the equation generally increased the significance of relationships observed, as sensitivity to hydrological connectivity will vary depending upon what the channel is connecting to, just as the sensitivity to land cover will vary upon the level of hydrological connectivity. This raises the point that it is neither land cover nor hydrological connectivity alone that determines in-stream conditions rather it is the interaction between the two factors that is important.

Table 5.7: *Comparison of the SCIMAP delivery index central tendency (un-weighted and weighted by land cover) for sites with trout fry present and sites with trout fry absent. Two-tailed Mann-Whitney tests were used.*

Year	Mann Whitney p value	
	Risk of hydrological connectivity	Risk of hydrological connectivity weighted by land cover
2002	0.0004	<0.0001
2003	0.003	<0.0001
2004	0.002	0.003
2005	0.476	0.089
2006	<0.0001	<0.0001

Table 5.8: *Comparison of Spearman Rank correlation results between the SCIMAP delivery index (un-weighted and weighted by land cover) and concentrations of potassium, phosphate, and nitrate. Tests were two-tailed.*

	Risk of hydrological connectivity			Risk of hydrological connectivity weighted by land cover		
Delivery Index	**Potassium**	**Phosphate**	**Nitrate**	**Potassium**	**Phosphate**	**Nitrate**
Correlation coefficient	0.696	0.443	0.411	0.757	0.453	0.301
p value	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001

Based on these results it is important to recognise that hydrological connectivity is neither inherently good nor inherently bad. Instead, its impact depends upon the context in which connectivity is applied, and which landscape parameter is being considered to connect with which in-stream ecological function. For example, in the case of fine sediment delivery, high hydrological connectivity between areas of the landscape with high erosion risk and channel reaches where salmonid redds are found may be considered "bad". However, in the case of nutrient delivery, high hydrological connectivity between areas of the landscape with low nutrient status and reaches where juvenile salmonids are feeding may be considered "good". Acknowledgement of this concept requires a shift in thinking about the term "connectivity" which particularly in ecology and fisheries science has become considered an inherently "good" characteristic (e.g. Moilanen and Hanski, 2001; Calles and Greenberg, 2005).

This research emphasises the importance of considering hydrological connectivity when evaluating the relationship between in-stream ecology and landscape parameters. As an explanation of the delivery mechanism for transporting sediment and nutrients to the channel network, the inclusion of hydrological connectivity within the *SCIMAP* framework and integration of this effect through to the river network adds functional significance to the relationship observed between land cover and salmonid abundance. Evidence supporting these mechanisms was provided by the relationships observed between the delivery index, gravel siltation and water chemistry. Adding this functional explanation is important for fisheries managers as it suggests why land cover may be important in structuring salmonid abundance (Poff, 1997). It also provides a mechanism for manipulating land cover impacts through the use of strategies such as buffer strips, wetlands, hedge planting, and beetle banks which can reduce the level of hydrological connectivity (dependent upon contaminant) in areas where excessive delivery is a problem (Burt, 2001). Alternatively, in areas where reduced delivery and transfer between the catchment and channel is a problem increasing hydrological connectivity, for example, through the removal of non-essential flood defences may be beneficial. Consideration of hydrological connectivity and the delivery of material to the channel may also explain differing views over the role of land use and land management in influencing ecology (Meador and Goldstein, 2003). It may also explain why changes in land management do not always achieve the desired results. Instead promoting consideration of landscape sensitivity and the degree to which the landscape is responsive or resistant to change (e.g. Brunsden and Thornes, 1979; Burt, 2001; Burt and Pinay, 2005) For example, targeting the application of nutrient budgeting and soil conservation techniques in sensitive areas of high connectivity is likely to have a greater impact on water quality than applying it in areas of low connectivity and low sensitivity.

One of the most interesting findings was the different response of salmon and trout fry at low values of the delivery index. This supports Hypothesis (2) formulated within Chapter Two of this thesis which stated that relationships between habitat and salmonid abundance are species and location specific. In this case it was suggested that their different response reflected their different feeding habits, with salmon fry more dependent on autochthonous production than trout fry. This finding has important implications for fisheries managers as different management approaches may be required at different locations within the catchment, dependent upon species. The *SCIMAP* delivery index and level of hydrological connectivity may also be a useful tool for explaining or predicting salmonid abundance and distribution in pristine environments. For example, reaches with a very low delivery index and hence low autochthonous food availability may simply not be suitable for supporting salmon populations naturally. In such areas implementing strategies such as habitat creation is likely to be ineffective at increasing salmon abundance as the limiting factor may not be habitat but food supplies. The delivery index may also provide a possible framework for explaining and predicting spatial patterns in growth rates and considering the migratory behaviour of salmonids. However, further research would be required to evaluate the potential for these applications.

5.8 Chapter summary

The aim of this chapter was to evaluate and to validate the potential of the *SCIMAP* model as a tool for assessing and quantifying the impact of catchment-scale controls on salmonid habitat. In particular, it aimed to assess the role of hydrological connectivity in structuring the relationship between salmonid abundance and land cover at the catchment-scale. This was achieved by relating the risk of surface hydrological connectivity weighted by land cover to data on salmonid abundance. Results have been presented indicating that salmon and trout populations within the Eden catchment are significantly structured by land cover as filtered by topographically-controlled hydrological connectivity. However, the impact of hydrological connectivity was found to be species-specific supporting Hypothesis (2) of this thesis. Mechanisms relating to the delivery of fine sediment, nutrients and feeding habits have been invoked to explain these results and are supported by the relationships observed between the risk of hydrological connectivity, gravel siltation and water chemistry. Overall, these findings support the use of *SCIMAP* as an ecologically relevant and appropriate tool for quantifying the impact of catchment-scale controls in this case land cover on salmonid habitat. The importance of these catchment-scale controls as compared with the riparian and in-stream controls discussed in Chapter Four will be assessed in Chapter Six of this thesis using multivariate statistical analysis to investigate the hypotheses identified in Chapter Two.

Chapter Six - Integration and assessment of relationships between habitat and salmonids: A hierarchical approach

6.1 Introduction

The aim of this thesis is to couple remote sensing, Geographical Information Systems (GIS), environmental modelling and ecological surveying techniques with current ecological understanding of habitat controls on salmonid populations, in order to develop a more effective approach to prioritising habitat restoration. To this end, Chapter Two reviewed and synthesised current understanding of in-stream, riparian and catchment-scale controls on salmonid habitat using a DPSIR framework. Based on this review, a number of controls relevant to salmonid habitat in the Eden catchment, a predominantly rural agricultural landscape, were selected for analysis as presented in Table 3.1. Three hypotheses regarding the relationship between these controls and salmonid populations were proposed as follows:

Hypothesis (1): Relationships between habitat and salmonid abundance are structured by life-stage according to the level of mobility and potential for dispersal at each life-stage.
Hypothesis (2): Relationships between habitat and salmonid abundance are species and location specific relating to the scale of habitat occupied by different species.
Hypothesis (3): The scale of analysis (e.g. catchment, sub-catchment, tributary) will influence the relationships identified between habitat controls and salmonid abundance, or alternatively the scale of the control will be related to the scale of its impact.

Chapters Three, Four and Five then focused on identifying, developing and validating tools for quantifying and assessing salmonid habitat, appropriate to each habitat control and scale of control. These three chapters applied a range of tools including aerial photography, DTM processing, satellite remote sensing, and the *SCIMAP* diffuse pollution model to quantify the habitat controls selected in Table 3.1, which, with the exception of hydraulic conditions, were highly successful. The aim of this chapter and Chapter Seven is to deliver Objective (3) of the thesis by using the habitat and salmonid population data acquired to test the above hypotheses and to discuss the results in the context of approaches to prioritising habitat restoration. This chapter will focus on data analysis and presentation of results. Chapter Seven discusses the results in terms of the hypotheses proposed above and in the context of approaches to prioritising habitat restoration using the Eden catchment as a case study. This chapter is structured into two main sections, (1) database development and (2) data analysis.

6.2 Database development

A spatially-structured hierarchical database is developed to integrate the spatially distributed habitat data from a range of hierarchical scales (in-stream, riparian and catchment) with data on salmonid abundance. To enable testing of the three hypotheses, the representation and preservation of scale within the database is critical. Advances in GIS technology have made the creation of such databases feasible. Specifically designed to manipulate, store and analyse large quantities of spatial data, GIS has become a fundamental tool for natural resource managers and ecologists concerned with the analysis of spatially referenced data (Johnson, 1990). When combined with multivariate statistics, complex relationships can be investigated and interpreted within their spatial context. In terms of developing effective approaches to habitat restoration, GIS technology offers an additional advantage through its visualisation and spatial mapping capabilities. These enable visual material on landscape characteristics and spatial patterns to be produced, providing policy makers with a better visual perspective on the areas for which they are developing policies (Johnson and Gage, 1997). This provides an explanatory mechanism for easing the transfer of knowledge from scientific research to fisheries managers, landowners, project funders and local communities in a form that is easily interpreted and accessible. Throughout this thesis ArcGIS v9 has been used for quantifying data on habitat controls due to: (1) its ability to efficiently manipulate and handle both raster and vector data; (2) its powerful visualisation capabilities; and (3) its compatibility with Microsoft Excel. It will also be used in this chapter to develop the spatial database.

Prior to development of the GIS database there are a number of specific requirements which should be considered if the proposed hypotheses are to be successfully tested. First, habitat data must be related to salmonid population data in a manner that is appropriate to the life-stage in question, to enable testing of Hypothesis (1). For example, the presence of gravels is thought to be particularly important in determining habitat suitability for fry, whereas the presence of coarser substrate such as cobbles and boulders and the presence of pools may be more important in determining habitat suitability for parr. Due to their restricted dispersal capabilities, the proximity of spawning habitat is thought to be crucial to fry survival and the presence of gravels may be used as an indication of spawning habitat within the immediate vicinity of electrofishing sites. Parr production, on the other hand, may be limited by the extent of fry production within their spatial range. If few fry are produced, then it is unlikely there will be many parr. Second, salmonid abundance data must be recorded by species (i.e. Atlantic salmon or brown trout) to enable testing of Hypothesis (2). Again, habitat data must be related to salmonid population data in a manner that is appropriate to the species in question. For example, salmon will only be found below impassable barriers and consequently only sites below barriers will be included in analysis of salmon data, but sites both above and below barriers will be included in analysis of trout data. Third, data must be associated with the various hierarchical scales of structure that exist within the catchment (e.g. catchment, area, reach) to enable testing of Hypothesis (3). This will be done by recording which area (Section 1.3.1) and tributary each survey site falls within.

6.2.2 Salmonid population data

As discussed in Chapter Three (Section 3.3), salmonid population data have been provided by the Eden Rivers Trust and Environment Agency which both undertake annual electrofishing surveys. Salmonid abundance data must be recorded within GIS and analysed independently for each year of collection (Pess *et al.*, 2002). Temporal variation in the number of adult spawners between years (Figure 3.6) may impact upon the number of juveniles observed in any one year, independent of habitat. As this thesis is concerned with analysing spatial distribution and variation, spatial integrity in the data must be preserved and all analyses will be conducted in terms of individual years. The assumption has been made that, whilst absolute abundance may vary temporally, relative spatial patterns of salmonid abundance and distribution will remain fairly constant over time. This is supported by research by the Eden Rivers Trust (e.g. Dickson, 2004). In this regard, data from 2004 and 2005 have been selected for analysis as the majority of habitat variables were collected during these two years. Prior to 2004, the additional habitat variables such as gravel siltation and gravel presence recorded at the time of electrofishing were not collected. In 2006 the electrofishing survey focused on trout and only surveyed the smallest tributaries within the catchment. Unfortunately, aerial photography was not available for many of these sites. Two ArcGIS point shapefiles `Fish2004` and `Fish2005` were created recording species-specific fry abundance (number caught in 5 minutes), classified fry abundance according to Table 3.5 and fry presence/absence data for each site. Parr data provided by quantitative cluster electrofishing (Section 3.3.2) in both riffle and pool habitat were only collected for 54 sites in 2005 (16 fully quantitative and 38 cluster sites). These data were recorded in the point shapefile `Parr2005` as species-specific abundance (number of parr in $100m^2$) and presence/absence data. In addition to considering population controls at individual life stages it is also important to remember the connections between them and principles of stock recruitment (Section 2.4.2). Parr abundance at a site will in part be determined by fry productivity and the level of recruitment from fry to parr within the parr's spatial range. To take account of this within the multivariate analysis, a raster surface estimating relative fry productivity has been produced using the 2004 fry electrofishing data and GIS processing. Specifically, an inverse distance-weighted interpolation technique available within ArcGIS Spatial Analyst was applied to create the surface from the 2004 n5min^{-1} fry data. A single year's data were selected because, as noted above, temporal variation between years may confound the spatial signal in fry productivity. A search radius for interpolation of 2000 metres was selected to represent the spatial range of parr, which is suggested to be in the order of 1-10,000m (Armstrong *et al.*, 1998). 2000m was selected as an arbitrary distance representing a compromise between the spatial resolution of fry data and the spatial range of parr. One limitation of this method is that the search radius represents straight-line distance taking no account of channel sinuosity; as such the actual channel distance included may vary from site to site relative to channel planform. Further, the search radius was applied equally upstream and downstream despite research suggesting that parr will be more influenced by fry productivity upstream due to a predominantly downstream dispersion (Elliot, 1994). A power value of 2 was applied giving more weight to nearby fry sites than those further away based on the assumption that dispersing parr will occupy the first suitable and vacant territory they find thereby conserving energy and reducing vulnerability to predation (Einum and Nislow, 2005). Interpolation was undertaken separately for each species and for each of the seven sub-catchments for which parr data were available. Prior to interpolation a spatial buffer was created around each sub-catchment to prevent data from another nearby catchment being incorporated. Finally, the species-specific fry productivity surfaces for each sub-catchment were merged to create a single raster file (Figure 6.1). Whilst there may be limitations associated with this technique, it is only intended to be a rapid mechanism for approximating fry productivity at one site relative to another and not for calculating accurate stock and recruitment numbers.

6.2.3 Habitat data

In Chapter Three a number of perceived key controls over salmonid habitat were selected as potentially relevant to the Eden catchment (Table 3.1), a predominantly rural landscape with 90% under agricultural production (Mackay Consultants, 2003). Chapters Three, Four and Five then focused on identifying tools capable of quantifying these controls resulting in a suite of 24 habitat controls (anthropogenic and natural) representing the Eden catchment landscape (Table 6.1). These have been combined with fish controls characterising dependence on productivity at the preceding life-stage (e.g. fry production) to produce the final variable suite. Variables that are understood to be life-stage or species-specific will only be included in the relevant analyses as presented in Table 6.1.

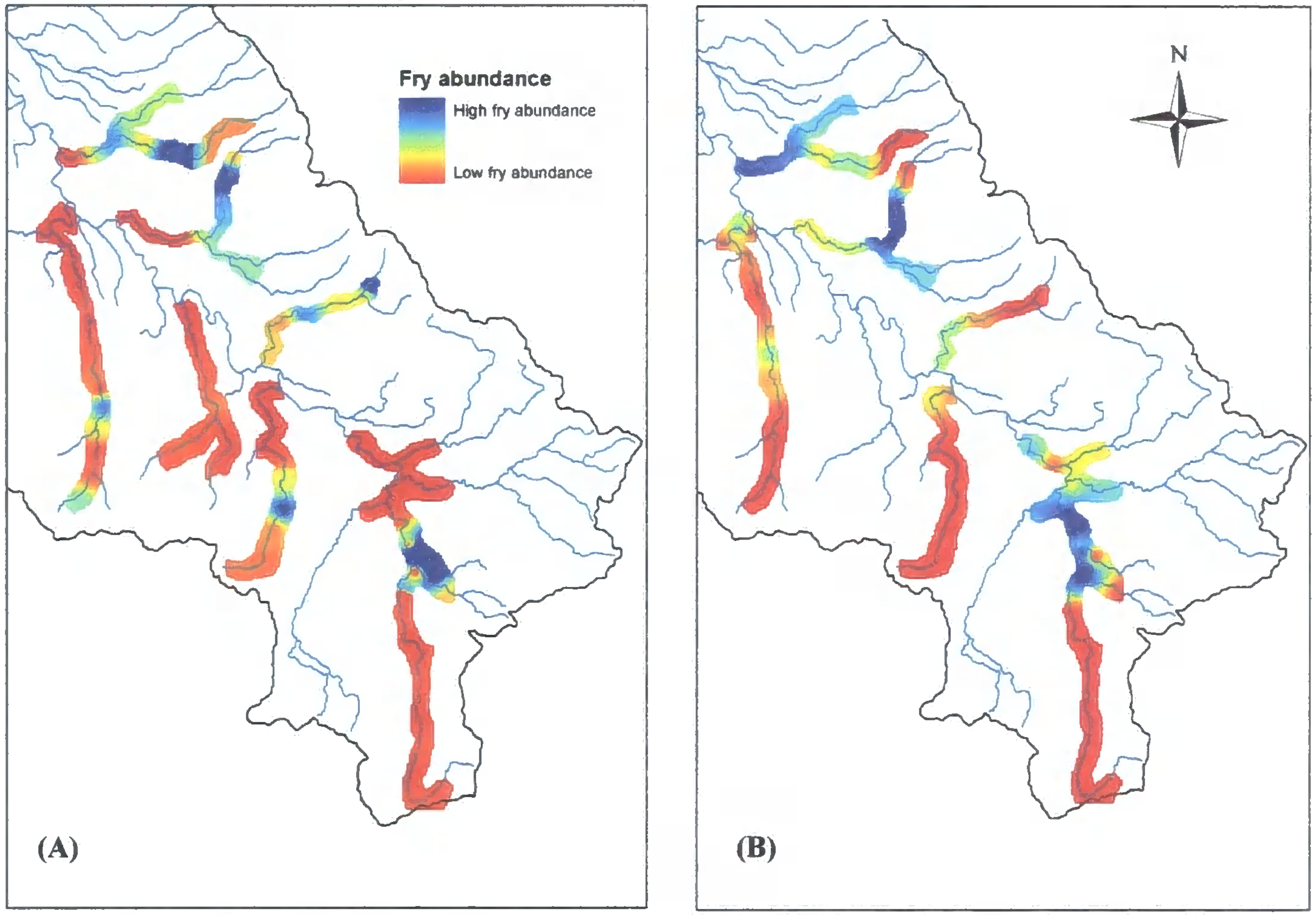


Figure 6.1: *Interpolated fry abundance (N5min^{-1}) for the Upper Eden catchment, (a) trout fry; (b) salmon fry. Interpolation was undertaken using electrofishing survey data provided by Eden Rivers Trust and the Inverse Distance Weighted (IDW) algorithm available in ArcGIS Spatial Analyst. A search radius of 2,000 metres was applied.*

Table 6.1: *Habitat and fish variables selected for inclusion in the spatial database.*

Variable	Source	GIS data type	Description and assumed impact based on expert opinion	Coding/Units	Included in salmon (S) and/or trout (T) analyses	Included in fry (F) and/or parr (P) analyses
Fish variables						
Trout fry abundance	EGIS	Raster	Trout fry production within trout parr range is assumed to affect parr production	Interpolated N5min^{-1} within 2,000m radius	T	P
Salmon fry abundance	EGIS	Raster	Salmon fry production within trout parr range is assumed to affect parr production	Interpolated N5min^{-1} within 2,000m radius	S	P
In-stream controls						
Dominant substrate	E	Point	Dominant substrate present within the site ranked according to assumed habitat suitability for fry. Class 3 most suitable, class 1 least suitable.	3 gravel/pebble 2 cobble/boulder 1 bedrock/sand/silt	S&T	F
Dominant substrate	E	Point	Dominant substrate present within the site ranked according to assumed habitat suitability for parr. Class 3 most suitable, class 1 least suitable.	3 cobble/boulder 2 gravel/pebble 1 bedrock/sand/silt	S&T	P
Gravel presence	E	Point	Gravel present, even if not dominant. It is assumed gravel is the preferred substrate for spawning and fry. It can be used as a proxy for proximity to spawning habitat.	1 present 0 absent	S&T	F
Gravel siltation	E	Point	Evidence of siltation within site. Silt is assumed to have a detrimental impact on habitat due to reduced oxygen levels within gravels and salmonid redds.	1 present 0 absent	S&T	F
Channel width	P	Point	Channel width is assumed to affect the ratio of bankside cover available relative to the wetted surface area. It also accounts for the scale of habitat occupied.	Metres	S&T	F&P
Impassable barrier	EA	Point	Sites upstream of impassable barriers (artificial or natural) to fish migration are assumed to have lower trout abundance and no salmon.	1 upstream of barrier 0 no barrier	T	F&P
Percentage pool	E	Point	The percentage of pool habitat within each electrofishing site is recorded. Pools are considered an important component of parr habitat.	Percentage of total area	S&T	P
Channel slope	D	Raster	Channel slope (%) at the site.	Percentage slope	S&T	F&P
Physical biotope	D	Raster	Flow types predicted from channel slope have been ranked according to their assumed habitat suitability for fry. 5 most suitable, 0 least suitable.	5 riffle 4 step pool 3 cascade 2 rapid 1 run 0 pool/glide	S&T	F

Table 6.1 cont....

Variable	Source	GIS data type	Description and assumed impact based on expert opinion	Coding/Units	Included in salmon (S) and/or trout (T) analyses	Included in fry (F) and/or parr (P) analyses
Riparian-scale controls						
Erosion presence	E	Point	Bank erosion of any type present within reach.	1 present 0 absent	S&T	F&P
Stock access	E	Point	Agricultural stock able to access the river from either or both banks. Stock access is assumed to have a detrimental impact through bank erosion & nutrient inputs.	1 access 0 no access	S&T	F&P
Erosion on both banks	P	Line	Presence of erosion on both, one or no banks. It is assumed that erosion on both banks is more detrimental than erosion on one bank or no erosion as a result of channel over-widening.	2 Erosion on both banks 1 Erosion on one bank 0 No erosion	S&T	F&P
Erosion severity	P	Line	Severe bank erosion is assumed to be more detrimental than non-severe erosion or no erosion.	2 Severe Erosion 1 Erosion but not severe 0 No erosion	S&T	F&P
Stock erosion	P	Line	Accelerated bank erosion due to stock bank trampling is assumed more detrimental to salmonids than other forms of fluvial or topographic erosion.	1 Stock erosion 0 No stock erosion	S&T	F&P
Fluvial erosion	P	Line	Bank erosion due to fluvial processes can result in bank undercutting providing cover or changes to channel morphology if severe and on both banks.	1 Fluvial erosion 0 No fluvial erosion	S&T	F&P
Overhead tree cover	P	Line	Percentage of channel cover provided by riparian trees. It is assumed that high levels of cover are detrimental to salmon due to reduced autochthonous production, but advantageous to trout due to increased cover	1 >90% 4 25-50% 2 75-90% 5 <25% 3 50-75%	S&T	F&P
Tunnelled vegetation	E	Point	Tunnelled vegetation with low levels of light penetrating the canopy is assumed to have a detrimental impact due to reduced autochthonous production.	1 tunnelled 0 not tunnelled	S&T	F&P
Riparian land cover	E	Point	Dominant adjacent land cover within the site ranked according to the degree of intensification, from 1 most intensive to 0 least intensive. It is assumed greater intensification will lead to reduced salmonid abundance.	0 other 0.05 moorland/woodland 0.3 extensive pasture 0.7 intensive pasture 1 arable	S&T	F&P

Table 6.1 cont…

Variable	Source	GIS data type	Description and assumed impact on fry counts based on expert opinion	Coding/Units	Included in salmon (S) and/or trout (T) analyses	Included in fry (F) and/or parr (P) analyses
Catchment-scale controls						
Catchment-channel hydrological connectivity risk						
Linear	S	Raster	Relative risk that the channel is connected to land in the upslope contributing area by surface flow pathways. Higher risk is assumed more detrimental.	Scaled between 0 (no risk) and 1 (high risk)	S&T	F&P
Classified linear	S	Raster	Relative catchment-channel connectivity risk re-classified into 3 classes of equal membership. Higher risk values are assumed more detrimental to fry.	1 <33.3% 2 33.3-66.6% 3 >66.6%	S&T	F&P
Non-linear	S	Raster	Relative catchment-channel connectivity risk re-classified into 5 classes of equal membership. It is assumed that both low and high levels of risk are detrimental and that the optimum level lies within the centre of the risk range.	1 40-60% 2 20-40% & 60-80% 3 <20% & >80%	S&T	F&P
Catchment-channel hydrological connectivity risk weighted by land cover						
Linear	S	Raster	Relative risk that the channel is connected to risky land covers in the upslope contributing area. Higher risk values are assumed more detrimental.	Scaled between 0 (no risk) and 1 (high risk)	S&T	F&P
Classified linear	S	Raster	Relative risk of connection to risky land covers re-classified into 3 classes of equal membership. Higher risk values are assumed more detrimental.	1 <33.3% 2 33.3-66.6% 3 >66.6%	S&T	F&P
Non-linear	S	Raster	Relative risk of connection to risky land covers re-classified into 5 classes of equal membership. It is assumed that both low and high levels of risk are detrimental and that the optimum level lies within the centre of the risk range.	1 40-60% 2 20-40% & 60-80% 3 <20% & >80%	S&T	F&P

Data Sources: (E) Eden Rivers Trust and Environment Agency electrofishing surveys; (EA) Environment Agency walkover survey; (S) *SCIMAP* diffuse pollution model; (D) NEXTMap Britain 5m digital terrain model processing; (P) 20cm digital aerial photography; (EGIS) GIS processing of Eden Rivers Trust electrofishing data

It should be remembered that there is uncertainty of varying degrees associated with the accuracy of all these habitat variables dependent on the method of collection. For example, the interpretation of erosion presence from aerial photography was estimated to have an overall accuracy of 78% when compared with ground truth data (Table 4.3), whilst variables produced using the *SCIMAP* model represent only the risk that a particular habitat condition is present at a point in the channel network. Even those variables that were collected by ground survey will be open to some degree of uncertainty due to the subjectivity involved in a surveyor's interpretation. In a number of cases, data on the same habitat control were available from both the aerial survey and ground observation at the time of electrofishing. To reduce uncertainty in the dataset the most accurate data sources were selected. For example, data on channel substrate and gravel presence were only obtained from ground observations as there was a high degree of uncertainty and many reaches of unknown substrate type associated with the data surveyed using aerial photography. In other cases, data from both sources has been included as they were quantified differently. For example, during the electrofishing survey, tree cover was only categorised as tunnelled or not tunnelled. During the virtual walkover survey, tree cover was classified into five categories, less than 25%, 25-50%, 50-75%. 75-90% and greater than 90% cover.

6.2.4 Integration and co-registration of data

As discussed in the previous three chapters, all the habitat variables have been derived within or converted to ArcGIS file formats. However, whilst all the variables are held within the same software package, they are not all directly comparable as they represent a mixture of point, line and raster data which, as Figure 6.2 illustrates, are not always spatially co-registered. Lack of co-registration could be due to a number of reasons including the accuracy of hydrological flow routing used to derive the channel network in raster datasets, accuracy in the rectification of aerial photographs and accuracy associated with the GPS recording of electrofishing site locations. To overcome this problem, and to facilitate multivariate analysis, the data have been co-registered and extracted to a single data format. ArcGIS processing was used to achieve this, and all variables were extracted to the point shapefiles `Fish2004`, `Fish2005` and `Parr2005` based on the location of the 2004 and 2005 electrofishing sites. These files already contained the salmonid population data and habitat data collected during the electrofishing surveys.

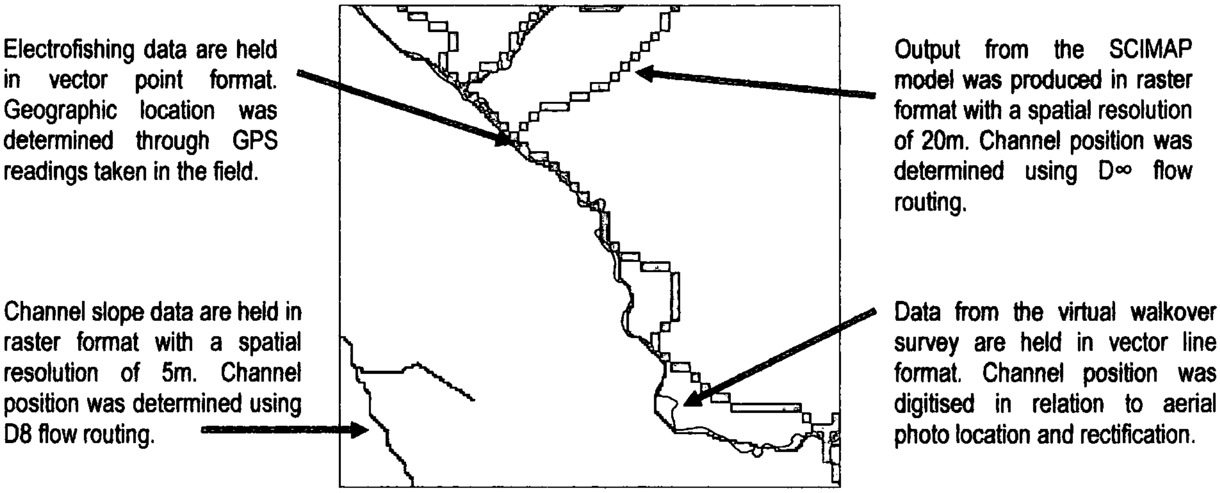


Figure 6.2: *GIS data type and issues of co-registration. Note that the different data sources to do not always overlie each other*

Extraction of data held in raster format

The risk of catchment-channel surface hydrological connection and risk of catchment-channel surface hydrological connection weighted by land cover, channel slope and physical biotope were all held in raster format. The position of the channel network in all cases was derived using GIS and topographically-based hydrological flow routing. The NEXTMap Britain™ 5m DTM and D8 flow routing algorithm were applied in the case of channel slope and physical biotope. A re-sampled 20m DTM and the D∞ algorithm were used in the case of the connectivity variables. In both cases, electrofishing sites frequently failed to overlie the derived channel network. To correct for this, and to enable data extraction, the following procedure was followed:

(1) Using the "*Reclassify*" function available in Spatial Analyst all cells in the raster layer relating to the channel network were classified as 1, with all other cells set to No Data.

(2) Using the "*Convert raster to feature*" function the raster cells with a value of 1 were converted to a polyline shapefile.

(3) The electrofishing sites held in point shapefiles were then snapped to the new channel polyline layer using a snapping tolerance of 200m, the aim being to move all points so that they overlay the raster dataset. Care was taken to check all points by referring to the electrofishing site name and site description in reference to digital map data to ensure wherever possible that the points had not snapped to the wrong stream or to a diagonal between two cells (Figure 6.3). Where this had happened or where points lay outside the

snapping tolerance they were manually moved using the functions available in the Editor Toolbar to overlie the correct raster cell.

(4) The extraction of data from each raster layer to each point was then automated using the "*Extract to point*" function available in the Spatial Analyst section of the ArcToolBox.

(A) (B)

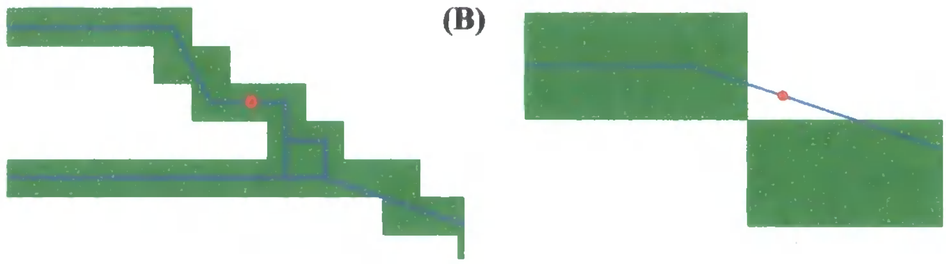


Figure 6.3: *Potential sources of error when snapping points to a polyline in GIS. (A) The point data may be snapped to an incorrect tributary. (B)The point data may not overlie a raster cell as a result of snapping to a diagonal between cells.*

Extraction of data held in vector line format

Variables obtained using aerial photography and the virtual walkover methodology were held as vector data in a polyline shapefile `RiparianHabitat`. In this case the position of the channel network had been determined by digitising a river centre-line in ArcGIS according to channel position in the rectified aerial photography. To extract data from this polyline to `Fish2004` and `Fish2005` the Spatial Adjustment Toolbar available in ArcGIS was used as follows:

(1) Fields for the five habitat variables (stock erosion, fluvial erosion, erosion severity, double bank erosion overhead tree cover) were created in the attribute table of the `Fish2004/ Fish2005/ Parr2005` (`Fish ****`) shapefiles.

(2) Using the "*Attribute transfer mapping*" function links were established between corresponding fields in the data source file `RiparianHabitat` and the target data file `Fish ****`.

(3) Attributes were then copied from `RiparianHabitat` to `Fish****` by first selecting the relevant channel reach (arc) containing the habitat data and then selecting the relevant electrofishing site (point) that data was to be transferred to. This was repeated for every electrofishing site in the dataset.

(4) At the same time the channel width at each site was also estimated from the aerial photography. It was decided to record this additional variable to aid testing of Hypothesis (2)

as channel width can be used to assess differences in the scale of habitat occupied by salmon and trout.

Extraction of data held in vector point format

Data on impassable barrier locations were held in vector format as a point shapefile (`Barrier`). By visualising both `Barrier` and `Fish****` in map format it was possible to manually code sites as above (coded 1) or below (coded 0) a barrier. A new field for the barrier variable was created within the attribute table of `Fish****`. Sites were then selected and labelled using tools available within the Editor Toolbar. A similar procedure was used to label sites according to the area and tributary in which they were found.

Again it is important to remember the issue of uncertainty associated with the accuracy of the habitat variables. The process of co-registration may have not always resulted in salmonid data being linked with the correct habitat variables. Attempts to control this source of uncertainty were made. For example, a relatively small snapping tolerance distance of 200m was used together with careful screening of snapped data. However, as the GIS processor was not the same person who undertook the electrofishing surveys, correct co-registration could not be guaranteed in all cases. Once the GIS dataset had been created a number of variables required summarising and recoding to reach the final format shown in Table 6.1. This was done in Microsoft Excel by exporting the attribute tables for `Fish2004`, `Fish2005`, and `Parr2005`.

6.3 Data analysis

An approach capable of interrogating large, multivariate datasets is identified and applied to relate habitat data to salmonid population data. As discussed in Chapter One (Section 1.4.3), a range of statistical methods have been applied within the scientific literature to undertake similar studies. For example, Stauffer *et al.* (2000) used a 2x2 factorial analysis of variance (ANOVA) to compare the influence of riparian cover (wooded, open) and watershed soil characteristics (high runoff potential, low runoff potential) on fish community composition in the Minnesota River Basin, USA. Canonical correspondence analysis was used by Wang *et al.* (2003) to relate a matrix of environmental predictor variables (e.g. watershed, reach and riparian variables) to a matrix of fish response variables (e.g. abundance, diversity, top carnivore %) to assess which scale of predictor variables explained the most variation in fish response. One technique that has been comparatively widely applied to compare suites of habitat variables to fish abundance is multiple regression analysis (e.g. Pess *et al.*, 2002; Walters *et al.*, 2003; Coley, 2003). For example, Pess *et al.* (2002) used multiple regression analysis to evaluate the relationship between landscape characteristics, land use and coho salmon *(Oncorhynchus kisutch*) adult spawner abundance at both a watershed and reach scale. As discussed in Chapter One (Section 1.4.3), the advantages of multiple regression analysis are that it can establish whether a set of independent variables explains a significant proportion of the variance in a dependent variable, and more importantly, establish the relative predictive importance of the independent variables (Garson, 2006). This is important as it allows identification of those controls that are mostly likely to be limiting salmonid populations, from which restoration strategies can be formed. For this reason multiple regression analysis is applied here. Also noted in Chapter One (Section 1.4.3), was the issue of spatial autocorrelation and collinearity within habitat data (Armstrong *et al.*, 2003). If correlated variables are independently included within the statistical analysis, their effects may be double-counted, resulting in selection of an end-model parameter suite that contains redundant parameters whilst not necessarily including those variables that exert most influence over population dynamics. This issue has been addressed in a number of ways within the scientific literature. Individual regression coefficients have be interpreted with caution only making inferences about the suite of habitat controls associated with fish abundance (Pess *et al.*, 2002). Alternatively, procedures such as Principal Components Analysis (PCA) have been used to screen variables, replacing them with transformed variables (factors) that are independent of each other but which still account for a significant proportion of the variance in the underlying habitat data (Walters *et al.*, 2003). PCA has the additional benefit of enabling relationships between habitat controls to be investigated and has therefore been selected here. By applying it at different scales (e.g. catchment and area), it will be possible to examine whether habitat controls relate to each other differently at different spatial scales and in different locations.

Data analysis is conducted in three stages. First, habitat controls are related to each other and salmonid fry data at a catchment-scale. This will help determine whether habitat explains a significant proportion of the spatial variation in salmonid abundance at the catchment-scale and, if so, which habitat variables explain the most variation. It will also enable relationships between in-stream habitat conditions and controls at riparian and catchment-scales to be examined as well as differences between Atlantic salmon and brown trout response to habitat. Second, habitat controls are related to each other and salmonid fry data at an area-scale. This will test whether relationships between habitat variables and between salmonid abundance and habitat remain constant or vary with changing spatial scale. Third, habitat controls are related to salmonid parr data for an area of the Upper Eden catchment. This will test the extent to which relationships between habitat and salmonid performance vary throughout the salmonid life-cycle. Results are presented and discussed in brief with more detailed discussion in terms of the three hypotheses, and approaches to restoration provided in Chapter Seven. All statistical analysis has been undertaken in SPSS v.12.

6.3.1 Catchment-scale analysis of habitat controls and salmonid fry

Sample sites were selected for inclusion in the data analysis according to the availability of all habitat variables at the site. 212 and 177 sites were therefore available for analysis in 2004 and 2005 respectively.

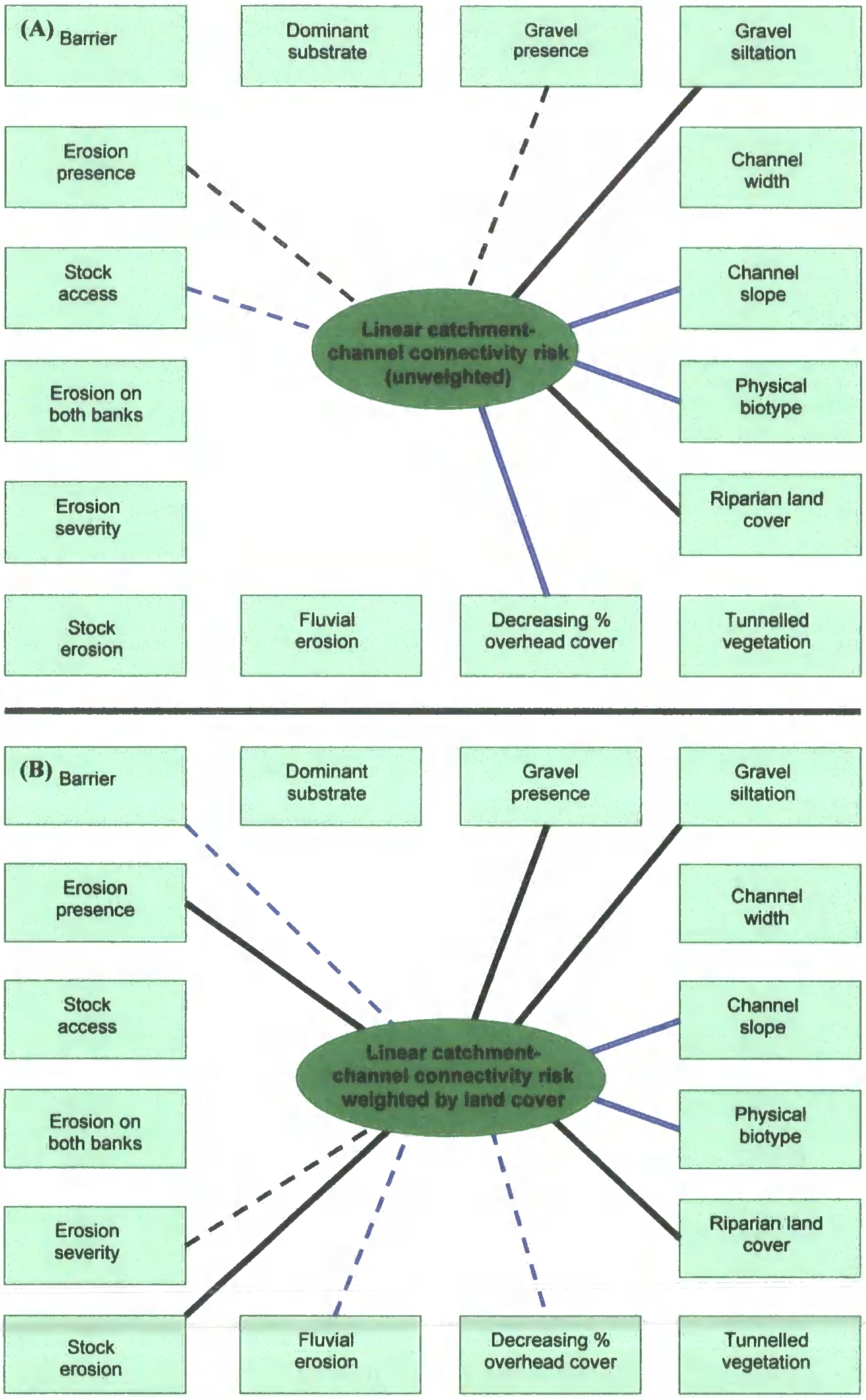
6.3.1.1 Correlation analysis of habitat controls

Initial investigation of the dataset focused on relationships between habitat controls using Spearman Rank correlation analysis applied to the combined habitat data for all 389 sites. As Table 6.2 indicates, there is a high level of collinearity within the dataset, with many variables correlating to each other both within and across scales. In particular, it is interesting to note the relationships observed between the catchment-scale variables and habitat controls operating at smaller spatial scales (Figures 6.4(a-d)). Considering these relationships provides an insight into the overall structure of the landscape, indicating that certain combinations of habitat controls are found to have a high probability of clustering together in certain locations of the landscape. For example, linear catchment-channel hydrological connectivity risk shows positive correlations with gravel siltation and riparian land cover at the 99% confidence level and with gravel presence and erosion presence at the 95% confidence level. Additionally, it exhibits negative correlations with channel slope, physical biotope and overhead cover at the 99% confidence level, and stock access at the 95% confidence level. This suggests that reaches experiencing high catchment-channel connectivity risk also have a high probability of experiencing gravel siltation, are typically found in areas of lower gradient where riparian land cover is typically more intensive and that there is a greater probability of overhead cover from riparian trees. Interestingly, the addition of the land cover weighting to linear hydrological connectivity risk increases the number of significant correlations found. For this variable, positive correlations are also observed with erosion severity and erosion due to stock and negative correlations are observed with erosion due to fluvial processes and the presence of impassable barriers.

Table 6.2: *Results of Spearman rank correlations between individual habitat controls. (*) Correlation is significant at the 0.05 level (2-tailed); (**) Correlation is significant at the 0.01 level (2-tailed).*

	Dominant substrate	Gravel presence	Gravel siltation	Erosion presence	Stock access	Tunnelled vegetation	Riparian land cover	Barrier	Channel slope	Physical biotope	Channel width	% overhead cover	Erosion on both banks	Erosion severity	Stock erosion	Fluvial erosion	Linear HCR	Classified linear HCR	Non-linear HCR	Linear weighted HCR	Classified linear weighted HCR	Non-linear weighted HCR
Dominant substrate		**	**	*			*	**	**	**												
Gravel presence	**		*					**	**								*			**	**	*
Gravel siltation	*	*		*	*		*	*	*	*	**				*		**	**		**	**	
Erosion presence	*		**		**	**	**		**		**	**	**	**	**	**	*	*		**	*	
Stock access			*	**		**	**					**	**	**	**	**	*	*				
Tunnelled vegetation				**	**							**	**	**		**		*				
Riparian land cover	*		*	**	**			**	**	**			**	**	**		**	**	*	**	**	**
Barrier	**	**					**		**			*							**	*	**	**
Channel slope	**	**	**	**			**	**		**	**	**		**	**		**	**		**	**	
Physical biotope	**		**				*		**		*						**	**	*	**	**	
Channel width			**	**					**	*			**									**
% overhead cover				**	**	**		*	**				**	**		**	**	**	**	*		
Erosion on both banks				**	**	**	**				**	**		**	**	**						
Erosion severity				**	**	**	**		**			**	**		**	**				*		
Stock erosion			*	**	**		**		**				**	**		**				**	**	
Fluvial erosion				**	**	**						**	**	**	**					*	*	
Linear HCR		*	**	*	*		**		**	**		**						**		**	**	**
Classified linear HCR			**	*	*	*	**		**	**		**					**			**	**	*
Non-linear HCR							*	**		*		**								*		**
Linear weighted HCR		**	**	**			**	*	**	**		*		*	**	*	**	**	*		**	**
Classified linear weighted HCR		**	**	*			**	**	**	**					*	*	**	**		**		**
Non-linear weighted HCR		*					**	**			**						**	*	**	**	**	

HCR = Hydrological connectivity risk


(A)
Barrier
Dominant substrate
Gravel presence
Gravel siltation
Erosion presence
Channel width
Stock access
Channel slope
Linear catchment-channel connectivity risk (unweighted)
Erosion on both banks
Physical biotype
Erosion severity
Riparian land cover
Stock erosion
Fluvial erosion
Decreasing % overhead cover
Tunnelled vegetation
(B)
Barrier
Dominant substrate
Gravel presence
Gravel siltation
Erosion presence
Channel width
Stock access
Channel slope
Linear catchment-channel connectivity risk weighted by land cover
Erosion on both banks
Physical biotype
Erosion severity
Riparian land cover
Stock erosion
Fluvial erosion
Decreasing % overhead cover
Tunnelled vegetation

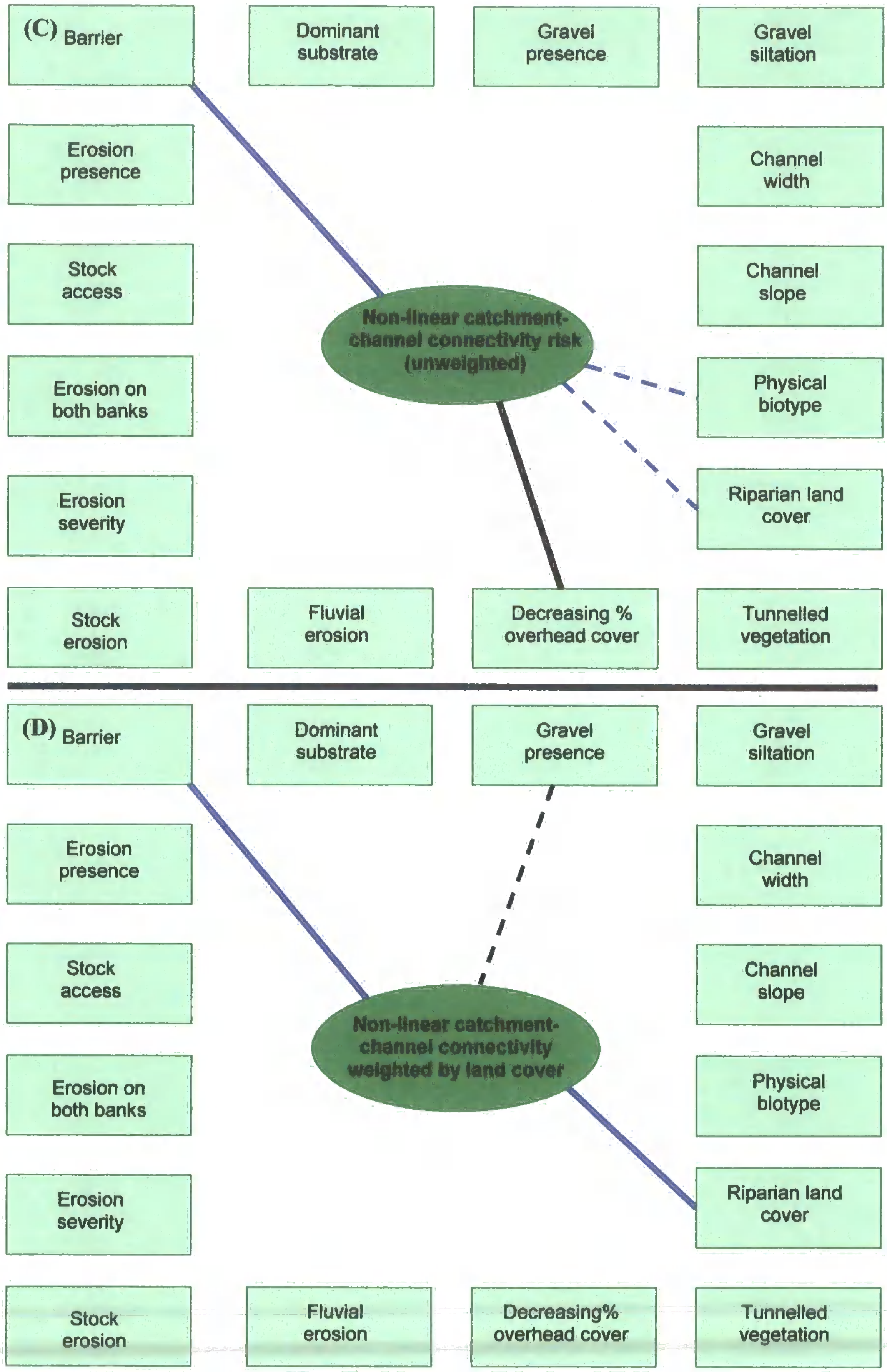


Figure 6.4: *Diagrammatic representation of Spearman Rank correlations between catchment-scale variables and habitat controls operating at smaller spatial scales. Solid lines represent correlations at the 99% confidence interval. Dashed lines represent correlations at the 95% confidence level. Black lines represent positive correlations and blue lines represent negative correlations. (Based on work by T. Burt)*

These results highlight the significant influence that land cover can have on the overall structure of the landscape resulting in multiple pressures on the freshwater environment at multiple scales. They also suggest that in specific locations of the landscape specific combinations of habitat pressures are likely to occur. For example, the correlations suggest that in lowland reaches (as represented by channel slope), there is a higher risk of hydrological connectivity between the river and land surface as controlled by the topography (gentle gradients and larger contributing areas generate a higher risk of soil saturation and overland flow), intensive agricultural land use, severe riparian damage due to intensive grazing and the siltation of gravels. However, these locations also have a higher probability of exhibiting positive habitat features such as the presence of gravels, overhead cover from riparian trees and being located downstream of impassable barriers. In upland reaches there is a greater probability of the opposite combination of habitat pressures occurring (Figure 6.5). These results emphasise the point that the impact of the environment upon salmonid habitat is complex with many factors combining to produce the final landscape and freshwater habitat structure within which salmonids exist.

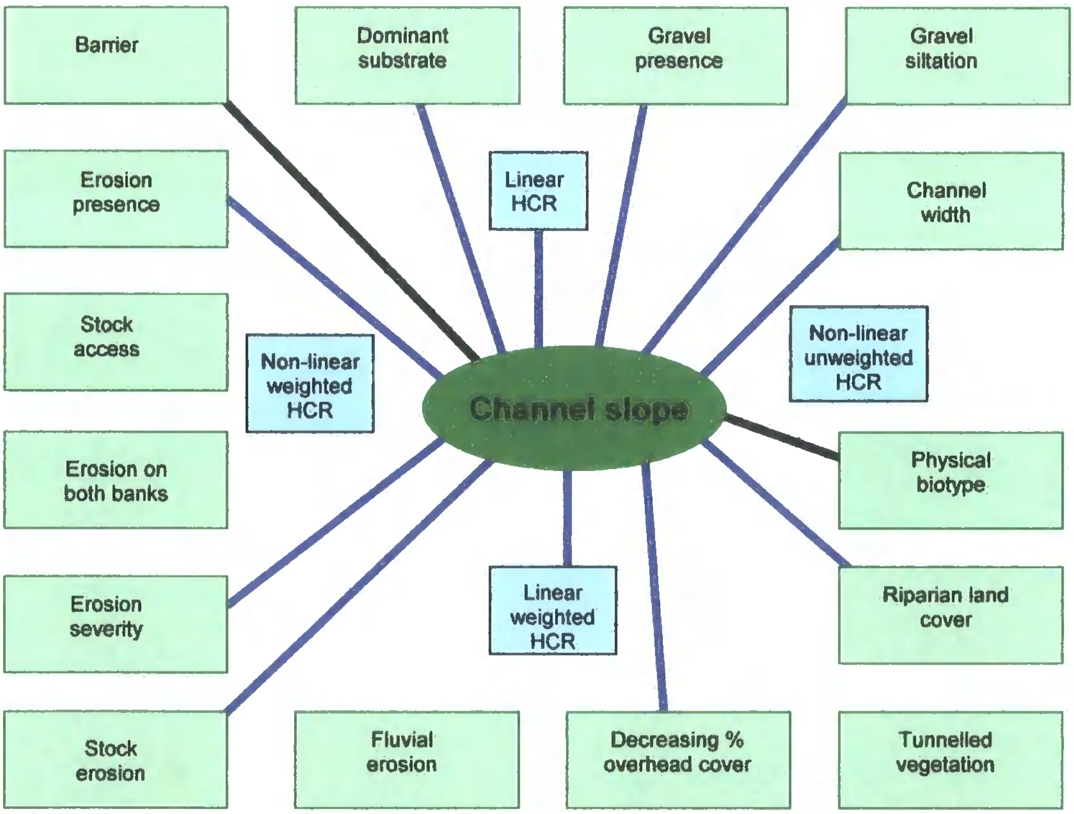


Figure 6.5: *Diagrammatic representation of Spearman Rank correlations between channel slope and other habitat controls. Black lines = positive correlations. Blue lines = negative correlations. All correlations are significant at the 99% confidence level.*

An additional point of interest is that far fewer correlations are observed between the non-linear connectivity risk variables (weighted and unweighted) and habitat controls at other scales (Figure 6.4c&d). This suggests that whilst salmon may respond to certain habitat controls in a non-linear fashion (Chapter Five, Section 5.4), relationships between hydrological connectivity weighted by land cover and other habitat controls are generally more linear in nature.

6.3.1.2 Principal Components Analysis

As discussed earlier, the presence of collinearity within datasets is a major issue to consider when undertaking multivariate analysis. This issue is to be addressed here by applying Principal Components Analysis (PCA) to create a suite of new factors that are independent of each other. In addition to producing independent factors, PCA also offers the opportunity to assess relationships between habitat controls in more detail. Unlike the salmonid data, the habitat data are considered to be relatively constant between years and data from sites in 2004 and 2005 were therefore combined to increase the sample size. PCA was applied with a varimax rotation to aid interpretation (Davies, 1984) and 7 factors with an eigenvalue greater than 1 were extracted accounting for 71.4% of the variability in the original dataset (Figure 6.6 and Table 6.3). All habitat variables selected for fry analysis (Table 6.1) were included with the exception of the barrier variable. This was excluded as it is only applicable to trout data, salmon sites only being found below barriers. Table 6.4 presents a summary of the habitat variables significantly represented by each of the factors and their relationship with those factors.

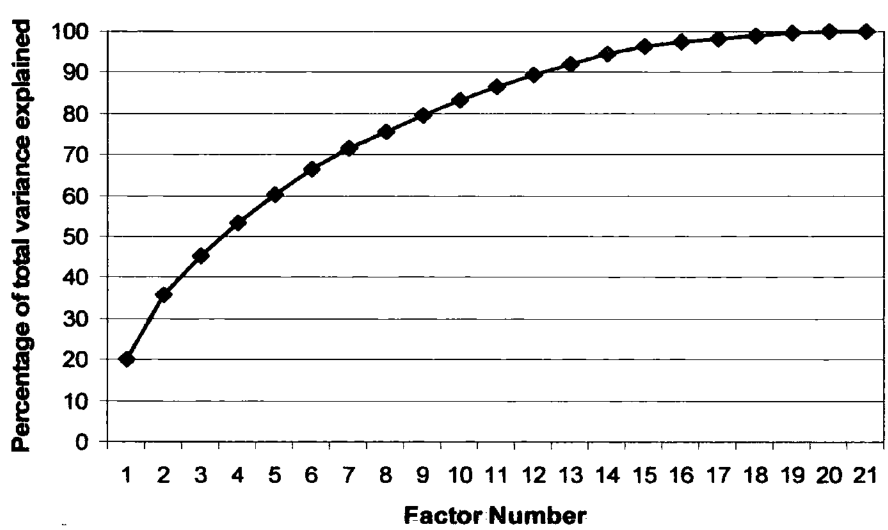


Figure 6.6: *Cumulative percentage of total variance in the original habitat dataset explained by the factors extract using Principal Components Analysis. The first seven factors have been selected for further analysis cumulatively accounting for 71.4% of the variance.*

Table 6.3: *Rotated component matrix of habitat factors extracted with an eigenvalue >1. The factor to which each habitat control is most strongly related is highlighted in bold.*

Habitat control	Factor Number 1	2	3	4	5	6	7
In-stream habitat controls							
Dominant substrate	-.034	.056	.007	.017	.157	.008	**.808**
Gravel presence	.137	.032	.066	.003	-.001	-.015	**.802**
Gravel siltation	**.362**	.144	.239	-.044	.123	.147	.200
Channel width	.010	-.228	-.338	.349	**.396**	.028	-.086
Channel slope	-.199	-.071	.087	-.082	**-.691**	-.137	-.174
Physical biotope	-.100	-.036	-.145	.066	**-.841**	.092	-.040
Riparian-scale habitat controls							
Erosion presence	.138	**.629**	-.026	.136	.015	.003	.138
Stock access	-.120	**.597**	-.017	.244	-.068	.194	.114
Erosion on both banks	-.044	**.890**	-.005	.059	.026	-.323	-.037
Erosion severity	.086	**.916**	-.051	-.002	.079	-.063	-.042
Stock erosion	.060	**.632**	.029	-.115	.081	.621	-.087
Fluvial erosion	-.077	.405	-.078	.170	.057	**-.838**	.002
% overhead channel cover	-.075	.193	.146	**.744**	.117	-.178	.016
Tunnelled vegetation	.051	-.137	-.027	**-.844**	.053	-.023	-.014
Riparian land cover	.165	.316	-.193	.134	.315	**.413**	.076
Catchment-scale habitat controls							
Catchment-channel hydrological connectivity risk							
Linear	**.957**	.004	-.129	-.044	.082	.011	.001
Classified linear	**.925**	.023	-.024	-.073	.040	-.033	-.043
Non-linear	-.012	-.085	**.901**	.096	.054	-.009	-.005
Catchment-channel hydrological connectivity risk weighted by land cover							
Linear	**.939**	.007	.065	-.012	.094	.069	.109
Classified linear	**.920**	.031	-.062	-.010	.121	.079	.035
Non-linear	-.053	-.055	**.910**	.060	-.052	.011	.067

Table 6.4: *Summary of habitat variables represented by factors extracted using Principal Components Analysis*

Factor	Name	Correlation	Habitat variables represented
1	Linear connectivity	Positive	Linear catchment-channel surface hydrological connectivity risk (weighted and unweighted by land cover) and gravel siltation
2	Bank erosion severity	Positive	Bank erosion, severe erosion, erosion on both banks, stock access and sock erosion
3	Non-linear connectivity	Positive	Non-linear catchment-channel hydrological connectivity risk (e.g. greater probability that risk falls outside the optimal range, Chapter 5, Section 5.4)
4	Overhead cover	Negative Positive	Tunnelled vegetation Decreasing percentage of overhead tree cover
5	Slope & biotope	Negative Positive	Channel slope and physical biotope Channel width
6	Fluvial erosion	Negative	Fluvial erosion
7	Gravel substrate	Positive	Dominant substrate suitability & gravel presence

Historically, fisheries managers have focused on in-stream habitat conditions often undertaking restoration projects aimed at treating the symptoms of habitat degradation rather than the causes (Summers *et al.*, 1996). Of particular concern has been the issue of gravel siltation and the infiltration of fines into salmonid redds (Soulsby *et al.*, 2001; Greig *et al.*, 2005; Crisp, 1996), often addressed through gravel cleaning programmes (Shackle *et al.*, 1999). However, unless the causes of siltation (e.g. bank erosion and/or catchment soil erosion) are addressed, gravel cleaning may well not be sustainable without ongoing maintenance. This thesis has stressed the importance of considering the influence of catchment and riparian-scale controls on in-stream conditions. Here PCA indicates that gravel siltation within the Eden catchment is positively related to both bank erosion severity (Factor 2) and catchment sources (Factors 1 and 3, risk of hydrological connectivity to land cover with a high risk of soil erosion), but that a stronger relationship was observed with Factors 1 and 3 than with Factor 2. Rotated component scores of 0.362, 0.239 and 0.144 were reported for Factor 1, Factor 3 and Factor 2 respectively. This suggests that at the catchment-scale, catchment sources of fine sediment may be more significant contributors to gravel siltation than bank erosion sources. However, some bank erosion sources still appear evident in the data and may be locally important (Figure 6.7). Some further sites appear to be related to both factors (indicated by overlapping ellipses). This has important implications for the type of restoration strategy that should be adopted to reduce siltation.

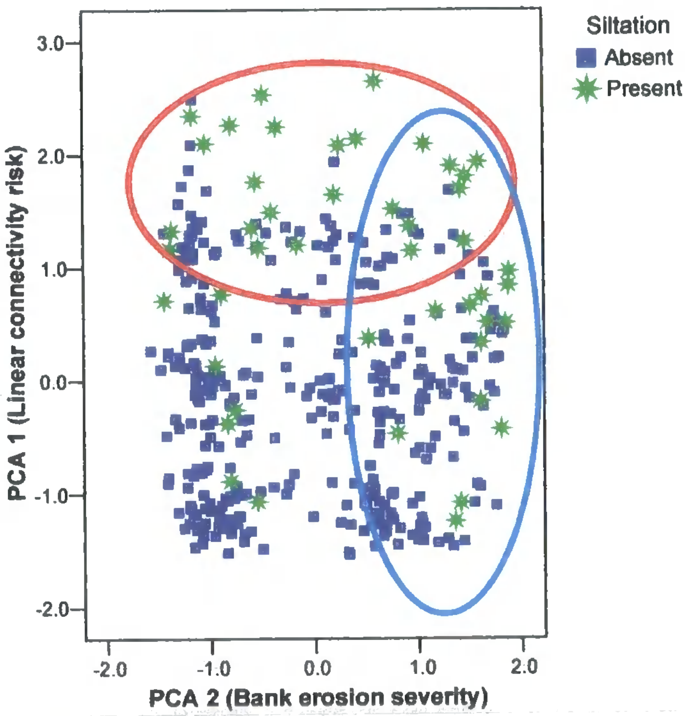


Figure 6.7: *Relationship between gravel siltation and fine sediment sources at a catchment-scale. Red ellipse indicates siltation associated with an increased risk of catchment soil erosion. Blue ellipse indicates siltation related to the presence of severe bank erosion. Placement of ellipses is arbitrary.*

It is also interesting to note that within the Eden catchment, Factor 2 (bank erosion severity), reports a higher rotated component score for erosion caused by stock grazing and bank trampling (0.632) than for erosion caused by fluvial processes (0.405). It is also positively correlated with increasing intensity of riparian land use (e.g. intensively managed pasture as opposed to extensively managed pasture). This suggests that stock access and increasing stocking levels may be a more significant cause of severe bank erosion at the catchment-scale than fluvial processes. Habitat controls also appear to be structured according to the scale of in-stream habitat as measured by channel width suggesting that the influence exerted by some controls may depend upon the scale of habitat considered. Although not scoring highly against any single factor, channel width does show moderate negative rotated component scores for Factor 2 (-0.228) and Factor 3 (-0.338) and moderate positive scores for Factor 4 (0.349) and Factor 5 (0.396). This suggests that narrow streams tend to have steeper channel slopes, and be more prone to severe bank erosion, channel widening (erosion on both banks), and extremes of catchment-channel hydrological connectivity risk, but that they have a higher percentage of cover from riparian vegetation and trees. Wider streams tend to have lower channel slopes and be less impacted by bank erosion and extremes of catchment-channel connectivity risk, but have a lower percentage of cover from riparian vegetation and trees.

6.3.1.3 Correlation analysis between habitat controls and salmonid populations

Initial investigation of the relationships between habitat controls and salmonid populations was undertaken using correlation analysis between individual habitat controls and trout and salmon fry abundance and presence/absence data for 2004. Prior to analysis the salmonid data were screened for the assumption of normality using the Kolmogorov-Smirnov test. However, even after application of square root and natural log plus 1 transformations the data were found to be non-normal, due to the high occurrence of zero and one fry sites (Figures 6.8 and 6.9). For this reason the non-parametric Spearman Rank correlation was applied (Table 6.5).

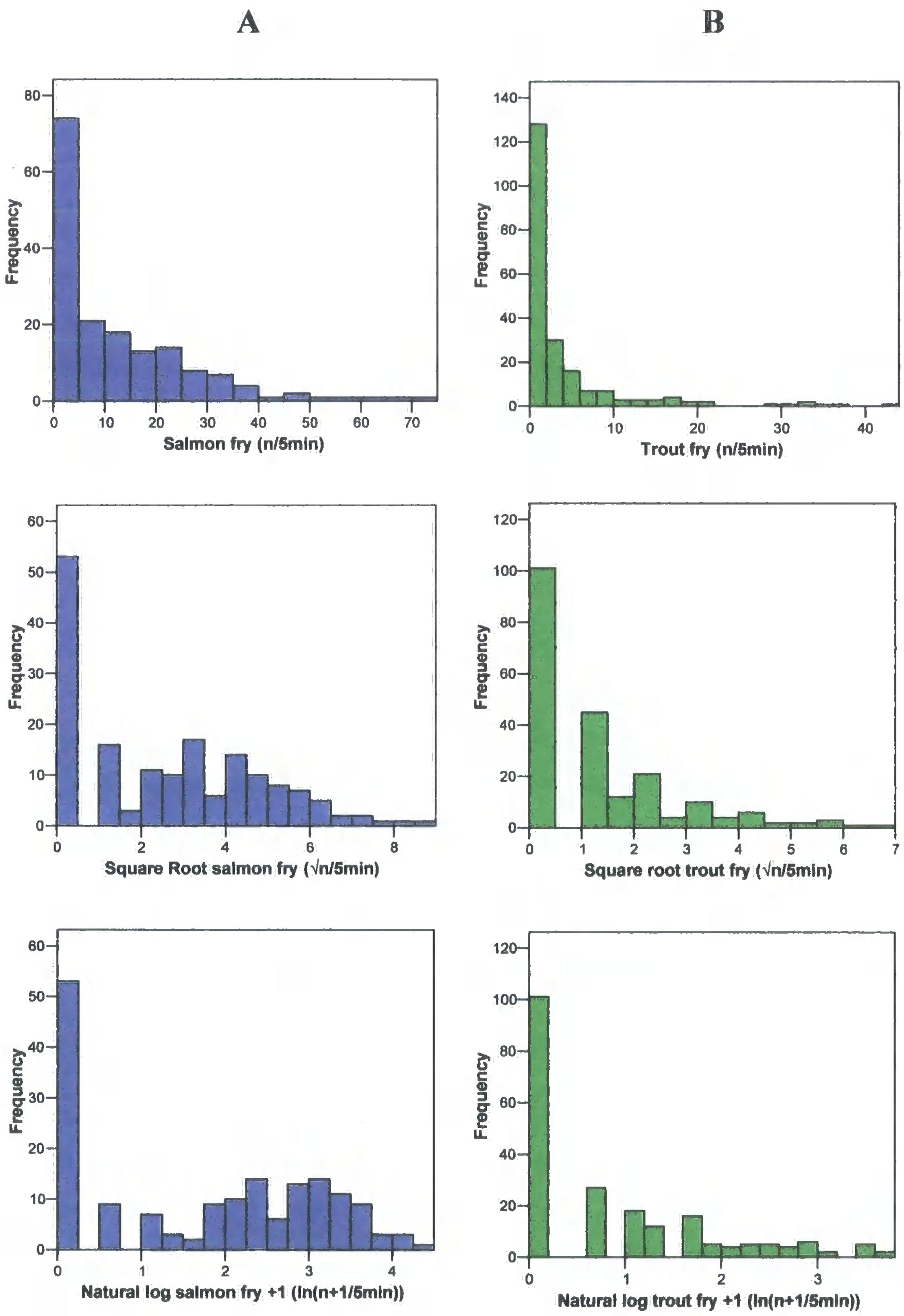


Figure 6.8: *Distribution of original and transformed fisheries data from 2004 (a) Atlantic salmon fry (b) trout fry.*

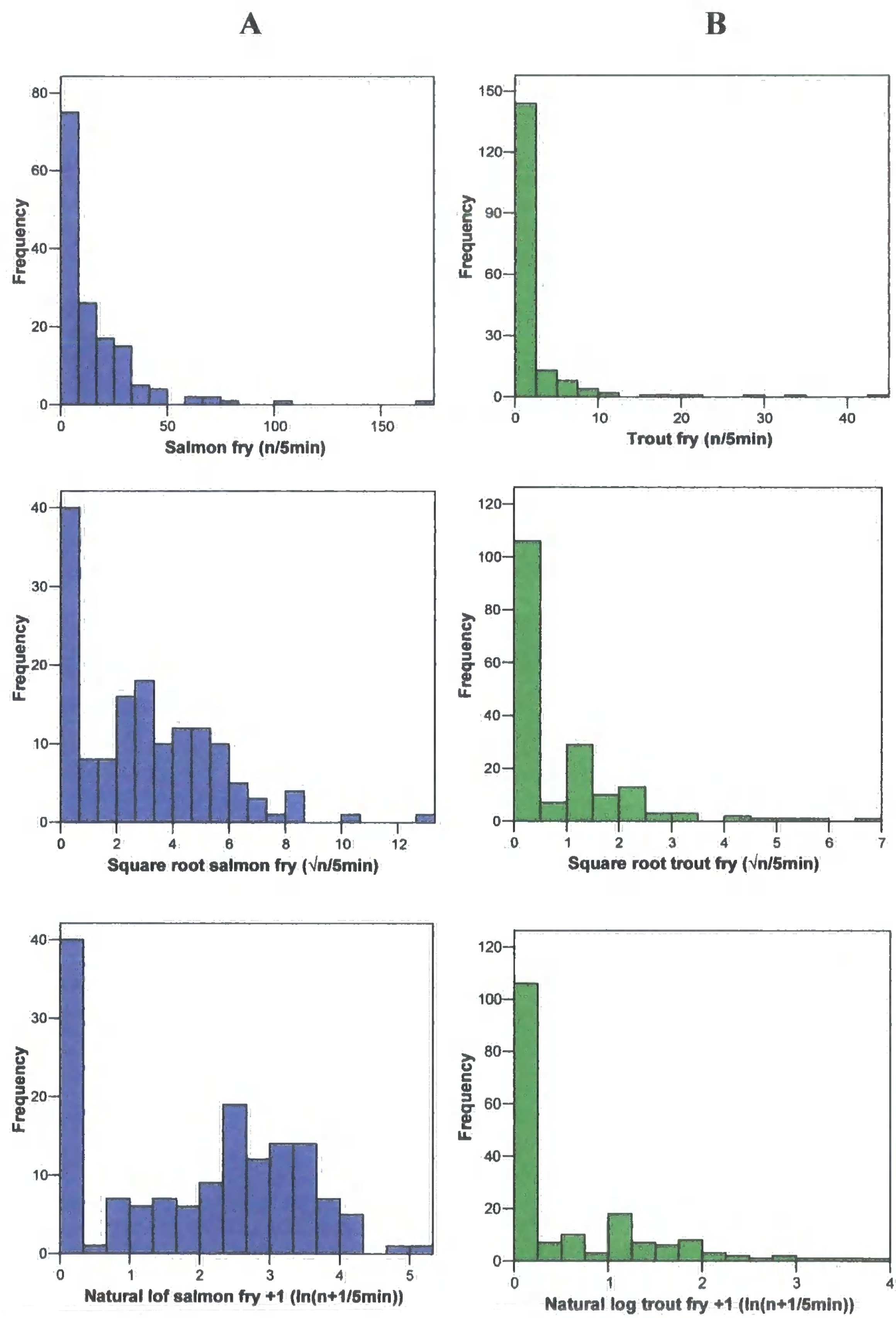


Figure 6.9: *Distribution of original and transformed fisheries data from 2005 (a) Atlantic salmon fry (b) trout fry.*

Table 6.5: *Correlation analysis between individual habitat controls and salmonid fry abundance and presence/absence at the catchment-scale. Spearman rank correlation was applied and all tests were two-tailed. Significant correlations at the 95% confidence level (p <0.05) are shown in bold.*

	Trout fry abundance n5min-1	Trout fry presence or absence	Salmon fry abundance n5min-1	Salmon fry presence or absence
In-stream habitat controls				
Dominant substrate	0.069	0.034	-0.004	-0.039
Gravel presence	0.013	-0.019	0.124	**0.136(*)**
Siltation of gravels	-0.044	-0.047	**-0.156(*)**	-0.119
Channel width	**-0.416(**)**	**-0.382(**)**	**0.374(**)**	**0.399(**)**
Barrier	-0.133	-0.036	**-0.477(**)**	**-0.524(**)**
Channel slope	**0.228(**)**	**0.251(**)**	**-0.261(**)**	**-0.231(**)**
Physical Biotope	**0.224(**)**	**0.224(**)**	**-0.136(*)**	-0.118
Riparian-scale habitat controls				
Erosion presence	**0.137(*)**	0.122	-0.030	-0.052
Stock access	-0.076	-0.044	-0.016	-0.050
Erosion on both banks	0.020	-0.008	-0.018	-0.047
Erosion severity	-0.033	-0.054	0.001	-0.037
Stock erosion	-0.022	-0.037	0.043	0.049
Fluvial erosion	-0.020	-0.032	0.002	-0.064
Tunnelled vegetation	**0.140(*)**	0.132	-0.053	-0.034
Overhead tree cover	**-0.176(**)**	**-0.140(*)**	0.028	0.020
Riparian land cover	-0.107	-0.109	**0.305(**)**	**0.265(**)**
Catchment-scale habitat controls				
Catchment-channel hydrological connectivity risk				
Linear	**-0.246(**)**	**-0.245(**)**	0.007	0.021
Classified linear	**-0.187(**)**	**-0.202(**)**	0.003	0.007
Non-linear	0.056	0.007	**-0.228(**)**	**-0.153(*)**
Catchment-channel hydrological connectivity risk weighted by land cover				
Linear	**-0.233(**)**	**-0.229(**)**	0.105	0.105
Classified linear	**-0.196(**)**	**0.207(**)**	**0.155(*)**	**0.165(*)**
Non-linear	0.125	0.093	**-0.316(**)**	**0.276(**)**

The results show that both salmon and trout fry exhibit a greater proportion of significant correlations with in-stream and catchment-scale controls than with riparian-scale controls. At the in-stream scale, significant correlations for trout fry included a negative correlation with increasing channel width and a positive correlation with increasing channel slope and associated physical biotopes. Conversely, salmon fry were positively correlated with increasing channel width but negatively correlated with increasing channel slope and physical biotope. In other words, trout fry indicate a preference for narrow streams with steep gradients whereas salmon fry indicate a preference for wider, low-gradient streams. This supports research suggesting that salmon and trout fry occupy different scales of habitat in different locations of the catchment (Armstrong *et al.*, 2003). In agreement, a negative Spearman Rank correlation between trout fry and salmon fry presence/absence ($p < 0.05$) was also observed. Salmon fry abundance was negatively correlated with the siltation of gravels and, unsurprisingly, with the presence of impassable barriers, whilst salmon presence was positively correlated with the presence of gravels. These three variables are considered to represent the quality, accessibility and availability of spawning habitat within the immediate vicinity of the surveyed fry habitat. The relationships observed between them and salmon fry suggest that salmon fry abundance and distribution at the catchment-scale may in part be controlled by spawning productivity and the level of survival to emergence. At the riparian scale only land cover was significantly positively related to salmon fry abundance and distribution suggesting salmon fry are found in locations of more intensive land cover (e.g. arable and improved pasture). This may correspond with their preference for wide low-gradient channels that are most likely to occur within lowland floodplains where the most intensive agriculture is also located. Trout fry abundance and distribution demonstrated slightly more significant relationships with riparian habitat controls being positively correlated with the presence of tunnelled vegetation and bank erosion and negatively correlated with a reduction in overhead tree cover. It is thought that these three variables represent the level of bankside cover available in the form of exposed tree roots, shade and undercut banks. At the catchment-scale, trout fry abundance and distribution were negatively correlated with increasing linear catchment-channel hydrological connectivity risk (and that weighted by land cover) indicating lower numbers of trout fry and even absence in areas of high connectivity risk. In contrast, salmon fry exhibited significant negative correlations with the non-linear catchment-channel hydrological connectivity variables in that both low and high levels of catchment-channel connectivity risk corresponded with low numbers of salmon fry, with an optimum level in the central range. This is in agreement with results presented in Chapter Five (Section 5.4) suggesting that salmon and trout exhibit a different response to low levels of connectivity potentially due to different feeding habits.

6.3.1.4 Regression analysis

Following correlation analysis with individual variables, multivariate analysis was undertaken by applying multiple regression to the derived PCA factors to examine which combination of habitat controls explained the most significant variation in salmonid fry populations at a catchment-scale.

Due to the non-normal distribution of fry, the data failed to meet the assumptions of normality required by linear regression even after transformation. To overcome this issue, the fry data were classified into 5 groups from A (Excellent) to E (Absent) and the regression model applied using ordinal dependents. This approach is considered allowable providing there are at least 5 response categories (Berry, 1993; Achen, 1991 *both cited in* Garson, 2006) and the responses are not concentrated in a very small number of those categories (Garson, 2006). The classification system adopted was that presented in Table 3.5 based on research by Crozier and Kennedy (1994) and Brown (2006a). Frequency statistics for each class, by species and year, were computed to check the assumption that data were not concentrated into a small number of classes. The assumption held for salmon fry but the large number of zero and one fish sites for trout meant that the assumption was not met. To adjust for this and to enable regression to be applied to the trout data the number of observations included in class E (absent) were reduced for both 2004 and 2005 using the process of random selection available in SPSS. It should be noted that salmon fry were only analysed for sites below impassable barriers due to the overriding influence of this factor on their distribution. 167 and 149 salmon sites and 148 and 97 trout sites were selected for analysis in 2004 and 2005 respectively. Figure 6.10 shows their distribution and the number of sites in each class is presented in Table 6.6.

Table 6.6: *Class frequencies used in regression analysis*

Class	Trout		Salmon	
	2004	2005	2004	2005
A	21	9	31	33
B	26	13	35	31
C	37	28	27	20
D	27	18	21	26
E	37	29	53	39

The 7 factors identified by PCA were then related to the ordinal classed fry data using forward stepwise multiple regression. Probability limits for variable entry into the model were set at 0.1, with variable removal set at 0.2. Previous research has advocated the use of 90% significance limits as at 95% fewer variables are retained which can lead to less informative results (Wang *et al.*, 2003). The barrier variable was also included in analysis of trout fry as it had not been included in the PCA analysis. Analysis was undertaken using data from all selected sites thereby evaluating the relationship between habitat and salmonid fry abundance at the catchment-scale. Multiple regression models were calculated separately for 2004 and 2005 and also for each species (Table 6.7).

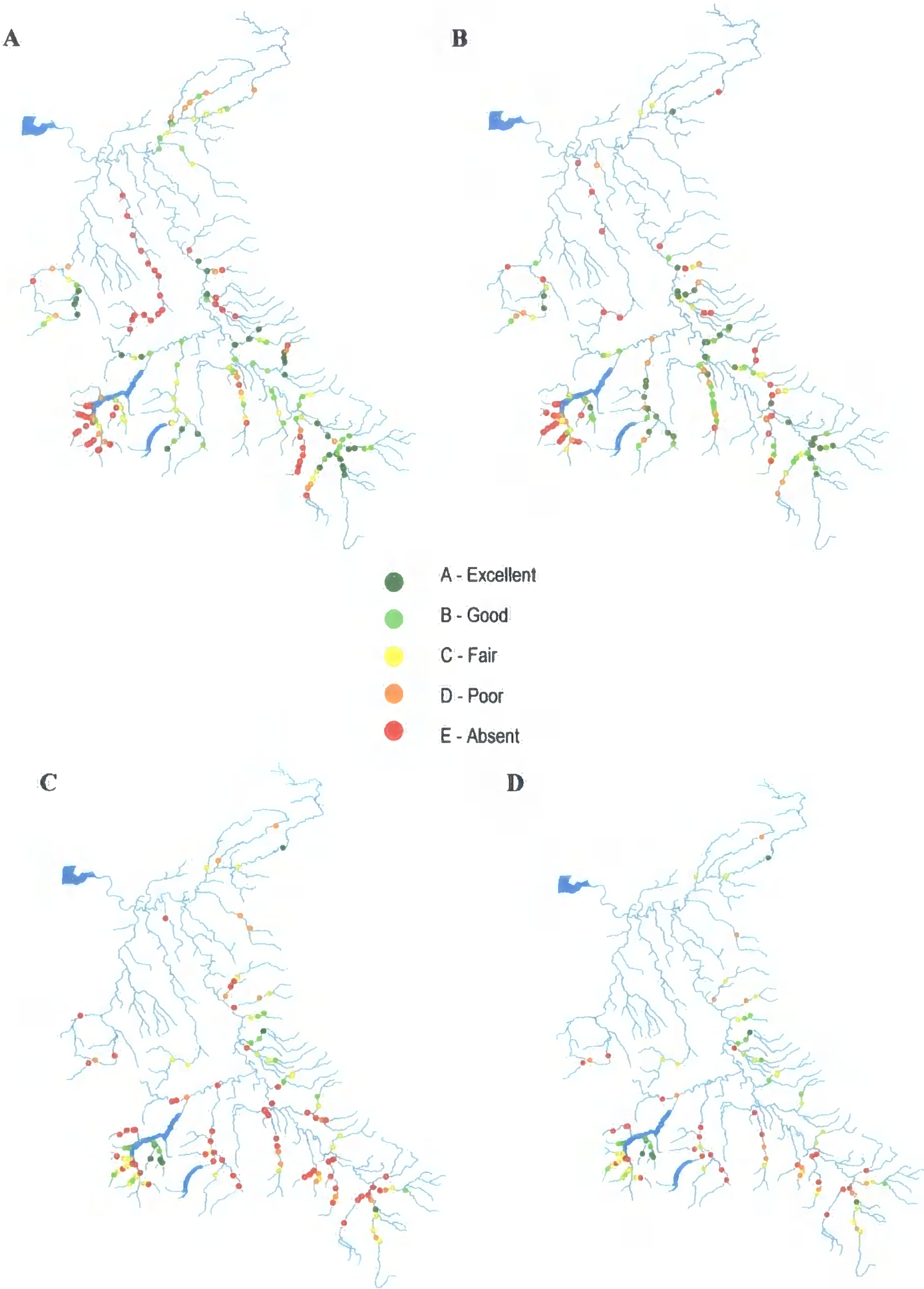


Figure 6.10: *Sample sites used in the multivariate analysis. (a) salmon fry 2004; (b) salmon fry 2005; (c) trout fry 2004; and (d) trout fry 2005.*

Table 6.7: *Multiple regression models summarising relationships between habitat variables and salmonid fry at the catchment-scale.*

Dependent	Regression equation	Adjusted R^2	Sample size
2004			
Salmon fry	= 1.870 - 0.737(Factor 3 Non-linear connectivity) + 0.362(Factor 5 Slope)	32.2%	167
Trout fry	= 1.878 – 0.506(Factor 4 Overhead cover)– 0.336(Factor 1 Linear connectivity) – 0.383(Factor 5 Slope) – 0.799(Barrier)	24.6%	148
2005			
Salmon fry	= 2.002 – 0.477(Factor 3 Non-linear connectivity) + 0.438(Factor 5 Slope)	17.8%	149
Trout fry	= 2.553 + 0.358 (Factor 4 Overhead cover) – 0.353(Factor 5 Slope) – 0.804 (Barrier)	15.0%	97

Adjusted R^2 values indicate that between 15.0% and 32.2% of the spatial variability present in the salmonid fry data was explained by the habitat controls at the catchment-scale. Whilst these adjusted R^2 values may appear low, in the context of a complex ecological system that has many small-scale interactions and other controlling factors, they are acceptable and equate to values observed by other researches (e.g. Pess *et al.*, 2002). They are also substantial considering the level of spatial noise that is likely to be associated with semi-quantitative (single pass, un-netted) electrofishing data (Wiley *et al.*, 1997). More variation was explained for salmon fry than trout fry with a greater proportion of the variation explained for both species, in 2004 compared with 2005. Three hypotheses are put forward to explain the different species response. First, salmon fry are more abundant in the Eden catchment than trout fry (Dickson, 2004). It has been proposed that habitat probably has its strongest effects when the standing stock approaches the carrying capacity of the environment (Armstrong *et al.*, 2003). Second, salmon fry exhibit wider spatial distribution within the Eden catchment than trout fry (Dickson, 2004), and hence may show a greater relationship with habitat when analysed at the catchment-scale. In connection with this, the reduction in zero fish sites for trout in 2005 resulted in a markedly lower sample size. Third, the spatial variability of trout fry populations may not have been fully captured, as prior to 2006, the Eden Rivers Trust did not target very small tributaries (<2m wide). In 2006 these small streams were specifically targeted showing that they can often be highly productive trout fry streams (Brown, 2006b). Unfortunately, aerial photography was not available for these sites preventing the application of multivariate analysis. However, analysis of relationships between the trout fry data and hydrological connectivity risk (Chapter Five, Section 5.4) found the most significant relationships to be observed for the 2006 data. In terms of the reduced explanation of variability observed in 2005 compared with 2004 this may be a result of the January 2005 floods that occurred in the Eden catchment (See Chapter Five, Section 5.4). The independent variables applied in these models have been measured using a variety of different units and as a result the coefficients reported in Table 6.7 are not directly comparable. Alternatively, the t-statistic (Table 6.8) can help establish the relative importance of each variable in the model, and as a guide, the greater the t-statistic is above +2 or below -2, the greater the importance of a variable. This is also reflected in the *p* values reported.

Table 6.8: *t-statistics and p values for catchment-scale regression models*

Model	t statistic	p value
Salmon 2004		
Factor 3	-7.886	0.000
Factor 5	3.784	0.000
Salmon 2005		
Factor 3	-4.190	0.000
Factor 5	3.947	0.000
Trout 2004		
Factor 4	-5.014	0.000
Factor 1	-3.352	0.001
Factor 5	-3.622	0.000
Barrier	-3.080	0.002
Trout 2005		
Factor 4	-2.910	0.005
Factor 5	-2.711	0.008
Barrier	-2.487	0.015

In both years, salmon fry were found to exhibit the strongest relationship (t-statistic > 4) with Factor 3, non-linear catchment-channel connectivity risk (and that weighted by land cover), in that both low and high levels of catchment-channel hydrological connectivity risk corresponded with low levels of salmon fry, with an optimum level in the central range. Trout fry abundance was also found to correspond with catchment–channel hydrological connectivity variables. However, it was Factor 1 that was found to be significant in this case. This represents a linear relationship with catchment-channel hydrological connectivity risk (and that weighted by land cover); where the lowest levels of risk are associated with the highest numbers of trout fry. These findings again correspond with analysis undertaken in Chapter Five (Section 5.4) which suggested that juvenile salmon and trout may respond differently to low levels of hydrological connectivity due to differences in their feeding habits. Whilst connectivity was found to be the most significant factor in explaining salmon fry abundance at the catchment-scale, for trout a negative relationship with Factor 4 (t-statistic = -5.014 in 2004) was found to be more significant (increasing Factor 4 represents decreasing overhead cover). As discussed in Chapter Two (Section 2.5.2.2) it is commonly thought that juvenile trout are strongly influenced by the availability of cover due to their high levels of territorial behaviour (e.g. Heggenes *et al.*, 1999; Armstrong *et al.*, 2003; Elliott, 1994). Trout fry were also negatively related to Factor 5, indicating that they are more frequently located in narrow channels with steeper gradients and more rapid flow types (e.g. cascades/step pools). Conversely, salmon fry were positively correlated with Factor 5, which represents a preference for gentler gradients and their associated flow types (e.g. riffles/runs/pools) that are typically found further downstream where channels are wider. In the context of the correlation results (Section 6.3.1.1), this would suggest that salmon and trout fry may be subject to different combinations of habitat pressures due to their location within the catchment landscape and scale of habitat utilised. Additionally, for trout fry, impassable barriers were found to be a significant factor affecting their abundance at the catchment-scale. Whilst local in scale, barriers can have a widespread effect depending on their location within the catchment. They may act to fragment habitats so that reaches above and/or below may no longer contain the range of habitat types required to support the entire life-cycle. In such cases the populations in these reaches will gradually decline until they are no longer sustainable. Barriers can also prevent re-colonisation of species following pollution or disease events upstream of the barrier.

These relationships were not only restricted to fry abundance, but also observed for the presence/absence of fry. Binary logistic regression (BLR) was chosen to assess the relationship between habitat controls and the presence/absence of salmonid fry. This is a technique specifically designed to predict the outcome of a dichotomous dependent variable (i.e. presence or absence) based on a set of predictor variables. The principal benefit of using logistic regression is that it does not rely on the dependent salmonid fry data being normally distributed and avoids the problems encountered above. It also enables any differences between salmonid distribution and abundance to be identified. BLR was undertaken using a forward likelihood ratio selection of variables to be entered into the model. The same suite of predictor habitat factors was used as in the multiple linear regression analysis and again the probability limits of a variable being entered or removed to/from the model were set at 0.1 and 0.2 respectively. Without the need to reduce the number of zero trout sites for BLR, 167 and 149 salmon sites and 212 and 177 trout sites were available for analysis in 2004 and 2005 respectively. Binary logistic regression models were again calculated separately for 2004 and 2005 and also for each species (Table 6.9).

Table 6.9: *Binary logistic regression models summarising relationships between habitat controls and salmonid fry presence/absence at the catchment-scale.*

Dependent	Regression equation	Nagelkerke pseudo R^2	Sample size
2004			
Salmon fry	= 1.159 – 1.290(Factor 3 Non-linear connectivity) + 0.714(Factor 5 Slope)	0.380	167
Trout fry	= 1.407– 0.786(Factor 1 Linear connectivity) - 0.997(Factor 4 Overhead cover) – 0.674(Factor 5 Slope)	0.317	212
2005			
Salmon fry	= 1.360 – 0.785(Factor 3 Non-linear connectivity) + 0.730(Factor 5 Slope) -0.360(Factor 2 Bank erosion severity)	0.232	149
Trout fry	= 0.169 – 0.447(Factor 5 Slope) – 0.356 (Factor 4 Overhead cover)	0.089	177

As the true R^2 cannot be computed for BLR, the Nagelkerke pseudo R^2 was used to provide a measure of goodness of fit. Similar to the multiple linear regressions, more variation was explained for salmon fry than trout fry, with a greater proportion of the variation explained for both species in 2004 compared with 2005. Very similar relationships were observed between habitat and salmonid presence/absence as described for salmonid abundance. However, in 2005, salmon fry presence was also negatively related to severe bank erosion, whilst for trout fry presence no significant relationship was observed with the presence of impassable barriers or hydrological connectivity.

6.3.1.5 Summary of catchment-scale analysis

Correlation analysis revealed a high degree of collinearity between habitat controls demonstrating the multi-faceted influence that land use can have on the freshwater environment. Specific locations of the landscape were also found to be associated with specific combinations of habitat pressures. PCA then evaluated relationships between habitat controls further. Relationships between in-stream habitat conditions, in particular, gravel siltation and habitat controls at a catchment and riparian-scale were observed. Relationships between the scale of in-stream habitat, as measured by channel width, and other habitat controls were also observed, particularly with respect to overhead channel cover, bank erosion severity, channel widening (erosion on both banks) and extremes of catchment-channel hydrological connectivity risk. Regression analysis then demonstrated that habitat controls do explain a significant proportion of the spatial variability observed in salmonid fry abundance and presence/absence data. However, there were indications that the January 2005 floods may have dampened relationships between fry and habitat in 2005. Following Chapter Five's results, catchment land cover as filtered by surface and shallow sub-surface hydrological connectivity was reported as particularly important in structuring the spatial pattern of salmonid presence/absence and abundance within the Eden catchment. However, as in Chapter Five, species-specific differences in response to low levels of connectivity risk were observed. Spatial variation in salmon fry performance was most significantly explained by a non-linear relationship with catchment-channel hydrological connectivity. Spatial patterns of trout fry abundance were most significantly structured by the level of overhead cover provided by riparian vegetation. Differences in the location and scale of habitat (as measured by Factor 5 channel slope, biotope and width) occupied by salmon and trout fry were also found. In the context of the correlation analysis results this suggests that salmon and trout may be subject to different suites of habitat pressures due to the location they utilise within the catchment. It also supports the theory of niche separation between the two species (e.g. Riley *et al.*, 2006; Heggenes *et al.*, 1999) which suggests that juvenile trout may perform well in narrow streams where bankside shelter is abundant relative to the area of the stream bed, whilst salmon, which are less dependent upon overhead cover, may thrive in wider streams where the bankside has less influence and where they are free from competition with trout (Armstrong *et al.*, 2003).The relationship between species differences, location and scale of habitat occupied and habitat quality will be discussed in more detail in Chapter Seven.

6.3.2 Area scale analysis of salmonid fry

To test whether the relationships identified between habitat and salmonid abundance remained constant or varied according to the scale and location of investigation, further analysis has been undertaken at an area, and where, possible tributary-scale. As described in Chapter One (Section 1.3.1), the catchment has been divided into six distinctive areas based on geology, topography and land cover: (1) the Tyne Gap; (2) the Pennnine Becks; (3) the Orton and Howgill Fells; (4) Ullswater and the River Lowther Valley; (5) the Caldew and Petteril Rivers; and (6) the Upper River Eden. Tributary scale analysis has focused on four tributaries within Orton and Howgill Fells area (Helm Beck, Hoff Beck, River Lyvennet and Scandal Beck). To examine variation in the spatial distribution of habitat controls at different scales scatterplots of four PCA factors were produced stratified by area and tributary (Figures 6.11 and 6.12). Visual inspection of the scatterplots suggests that different locations within the catchment do exhibit different habitat characteristics to different extents, particularly in relation to PCA Factor 1 which represents land cover and the level of catchment-channel hydrological connectivity risk (linear). At an area scale the strong clustering of sites within the Ullswater and River Lowther area at very low values of

Factor 1 (e.g. low levels of catchment-channel connectivity risk) is probably most striking. Sites within the Upper River Eden area are also relatively clustered towards low values of Factor 1. Sites within the Tyne Gap, Pennine Becks and Orton and Howgill Fell areas are less clustered indicating that they experience a wider range of catchment-channel connectivity risk but few sites exhibit extreme values. Interestingly, the River Caldew/Petteril area shows a bimodal distribution with sites clustered at both very high and relatively low values of Factor 1. This probably represents a distinction between sites within the upper River Caldew that is more topographically akin to the upland becks of the Ullswater area and sites within the lower River Caldew and River Petteril that are found within a more lowland environment. Figure 6.12(a) shows that stratification of Factor 1 is even more pronounced at the tributary-scale. Whilst there is relatively high spatial variation in Factor 1 between tributaries there is very little variation within individual tributaries, with the possible exception of Helm Beck. In contrast Factors 2, 4 and 5 which represent riparian and in-stream habitat controls (e.g. bank erosion, channel cover and channel slope) appear to show very little change in the degree of spatial variance with changing investigation scale. In other words, there is still a high degree of scatter (high variance) within areas and tributaries and relatively little or no stratification between areas and tributaries.

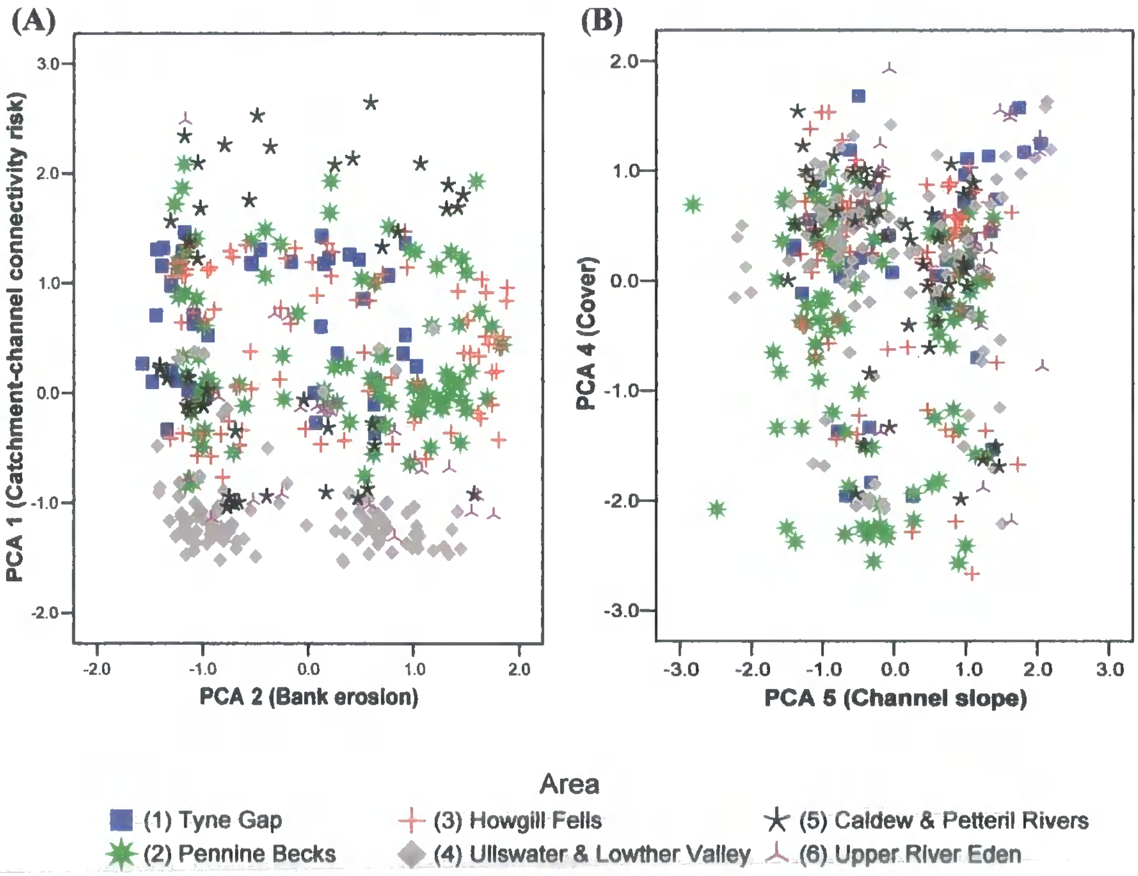


Figure 6.11: *Scatterplots highlighting the spatial distribution of habitat factors stratified by area.*

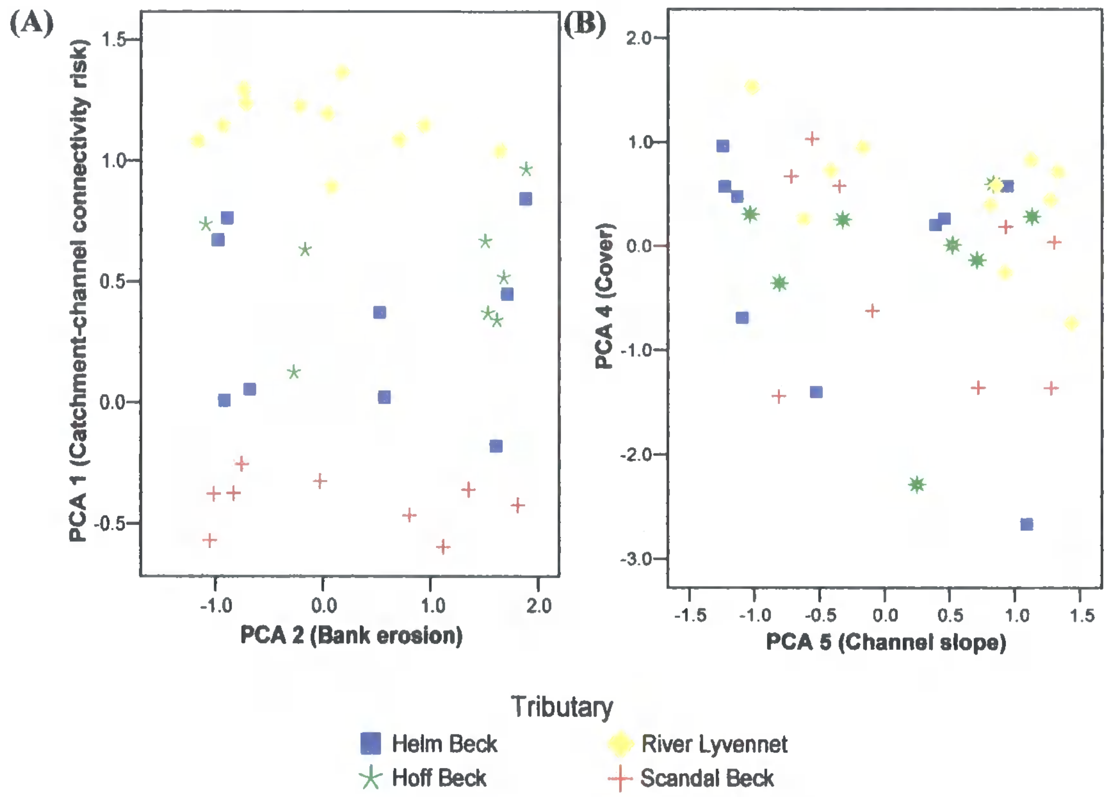


Figure 6.12: *Scatterplots highlighting the spatial distribution of habitat factors stratified by tributary.*

The contrasting behaviour between catchment-scale habitat controls and riparian/in-stream controls is clearly illustrated by Table 6.10. At the catchment-scale, the variance for all PCA factors equals one, as standardised by the extraction procedure. As the scale of investigation contracts to the area and then tributary-scale, there is a noticeable reduction in variance (with the exception of the Caldew/Petteril area) for Factors 1 and 3, which represent the catchment-scale influence of land cover and hydrological connectivity. This is the result of stratification in the factor score distribution between areas (Figure 6.13a). In other words, the factor scores for individual areas occupy a reduced proportion of the total factor score distribution expressed at the catchment-scale.

Table 6.10: *The impact of investigation scale on the level of spatial variance reported for each PCA habitat factor.*

Factor	Variance										
	Catchment scale	Area-scale						Tributary-scale			
		1	2	3	4	5	6	1	2	3	4
1	1	0.344	0.504	0.416	0.199	1.580	0.698	0.139	0.069	0.017	0.012
2	1	0.773	1.052	1.239	0.809	0.870	0.858	1.477	1.327	0.745	1.271
3	1	0.613	0.965	0.577	0.559	1.248	0.581	0.481	0.205	0.148	0.073
4	1	0.869	1.104	0.788	0.801	0.896	1.217	1.391	0.819	0.367	0.932
5	1	1.004	0.944	0.844	1.050	0.728	0.799	0.957	0.635	0.780	0.752
6	1	0.996	1.086	0.681	0.947	0.806	1.049	0.894	1.549	0.583	0.597
7	1	0.929	0.745	1.229	1.054	0.786	1.246	0.848	0.803	1.636	0.597

In contrast, there is little change in variance for Factors 2, 4, 6, and 7 which represent riparian and in-stream controls, as there is little stratification of factor scores between areas (Figure 6.13b). In other words, the factor scores for individual areas exhibit a similar distribution to that expressed at the catchment scale.

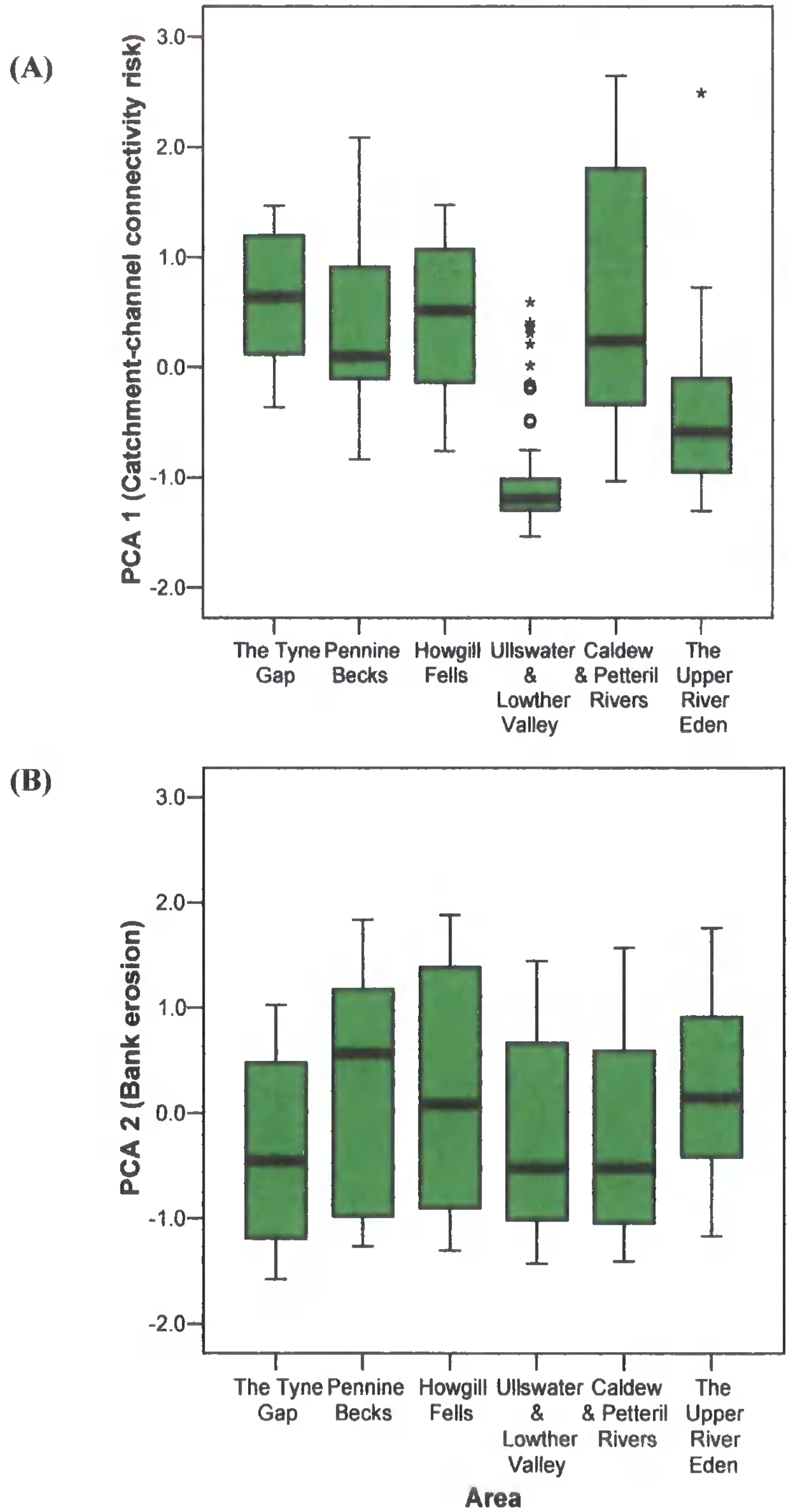


Figure 6.13: *Boxplots illustrating differences in the distribution of (a) PCA Factor 1 (catchment-scale land cover and hydrological connectivity) and (b) PCA Factor 2 (bank erosion severity)as stratified by spatial area.*

These results would suggest that, as the scale of investigation contracts, the level of spatial variation (habitat heterogeneity) exhibited by catchment-scale controls is reduced, whilst spatial heterogeneity in riparian and in-stream habitat controls remains relatively high. An alternative perspective is that catchment-scale land cover and connectivity exert their influence over a greater spatial extent relative to riparian and in-stream controls that are more localised in extent.

6.3.2.1 Principal components analysis

In addition to assessing the impact of investigation scale on individual habitat controls, consideration was given to the impact upon relationships between habitat controls. This was achieved by repeating PCA analysis specific to each area. Unfortunately, restrictions with the number of sample sites in area (6), the Upper River Eden, and in all four tributaries meant that it was not possible to apply PCA. PCA was applied with a varimax rotation and 6 factors with an eigenvalue greater than 1 were extracted within each area accounting for 70.8-77.9% of the variability in the original dataset. The rotated component matrices are presented in Appendix Four for information. Whilst a similar array of factors was produced for each area as at the catchment-scale, there were a number of interesting differences. First, relationships between the linear and non-linear risk of catchment-channel hydrological connectivity appear different at the area-scale compared with the catchment-scale. Instead of two different factors representing each variable they are all represented by one main factor in all areas. This suggests that whilst unrelated at a catchment-scale these two variables become related at the area-scale. Figure 6.14 shows that a positive relationship between linear and non-linear connectivity is observed for all areas except the Ullswater and Lowther Valley where a negative relationship is observed.

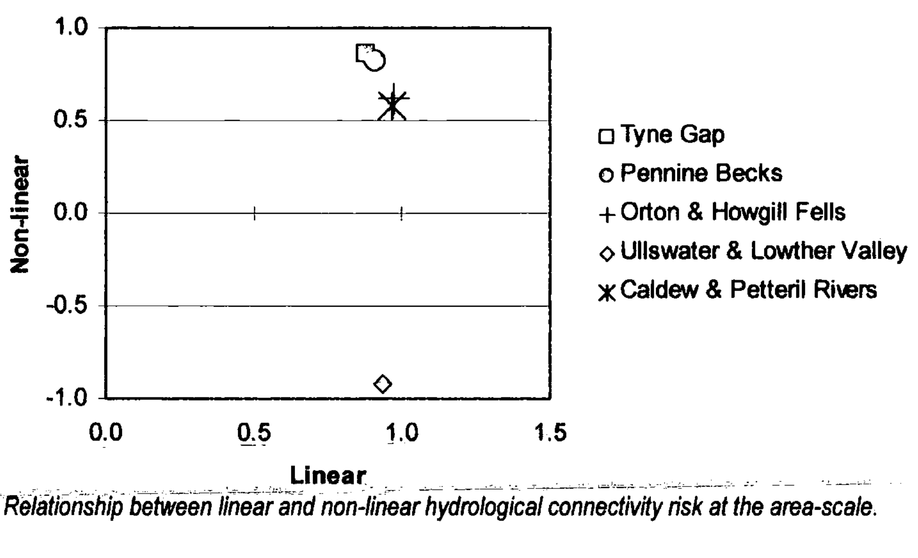


Figure 6.14: *Relationship between linear and non-linear hydrological connectivity risk at the area-scale.*

High values of the non-linear variable represent extremes in connectivity risk (both low and high), whilst high values of the linear variable represent only high levels of connectivity. A negative relationship between the two variables in the Ullswater area indicates that only extreme low levels to moderate levels of connectivity are found, whist a positive relationship in all other areas indicates that only moderate to extreme high levels of connectivity are found. These results again indicate a reduction in the spatial variability of catchment-channel connectivity risk experienced within areas as the scale of investigation contracts.

Second, the presence of gravel siltation is related to different factors in different areas. At a catchment-scale, siltation showed the greatest positive relationship with catchment-channel connectivity and land cover risk (as measured by rotated component scores). At an area-scale this is only true for the Caldew/Petteril Rivers area, which was also the area to exhibit greatest spatial variation in catchment-channel connectivity at the area-scale (Figure 6.13a). In the Ullswater/Lowther Valley area, siltation showed the greatest positive relationship with the presence of bank erosion caused by stock. This is the only area not to exhibit high levels of catchment-channel connectivity. In the Tyne Gap, Pennine Becks and Orton/Howgill Fell areas, siltation showed the greatest positive relationship with decreasing channel slope and associated low-turbulence physical biotopes. These results suggest that at different investigation scales, different controls may be relevant to explaining spatial variation in gravel siltation. This has important implications for practitioners who are looking to identify and manage the causes of habitat degradation. For the case of gravel siltation a hierarchical approach for assessing risk within the Eden catchment is proposed as follows:

(1) Spatial variation in siltation is firstly related to variation in the delivery of fine sediment from catchment sources as controlled by surface hydrological connectivity.

(2) In areas of low connectivity where delivery of fine sediment from catchment sources is low, spatial variation in siltation is related to variation in the location of bank erosion sources, as this becomes the dominant source.

(3) Within areas or reaches of relatively uniform fine sediment delivery risk (e.g. low spatial variability in catchment-channel connectivity risk or erosion risk, dependent on the dominant source), the presence of gravel siltation is related to channel slope and biotope. Channel slope and biotope can be considered surrogate variables for in-stream hydraulic properties

such as velocity, discharge and shear stress which control the transport or deposition, dispersion or accumulation of sediment.

Third, the PCA results also indicate that contrasting causes of erosion are related to bank erosion severity in different areas of the catchment (Figure 6.15).

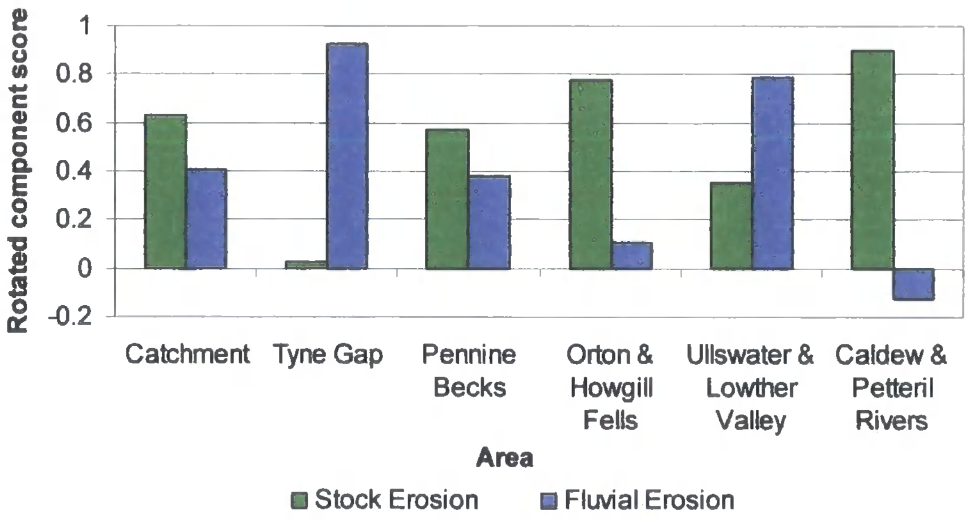


Figure 6.15: *Relationship between bank erosion severity and the cause of erosion. Rotated component scores are presented for the PCA factor in each area which represents bank erosion severity.*

At the catchment-scale, stock trampling showed a slightly greater association with severe bank erosion than fluvial processes. However, at the area-scale, whilst stock trampling still shows a greater relationship with erosion severity in the Pennine Becks, Orton/Howgill Fells and Caldew/Petteril areas, it is fluvial processes that appear more related to erosion severity in the Tyne Gap and Ullswater/Lowther Valley areas. This is likely to reflect differences in geology, topography and land cover between areas, and their respective controls upon flow regime, channel bed and bank material and stock access. In addition, differences in the relative importance of each erosion control become more marked at the area-scale especially for the Tyne Gap, Howgill Fells and Caldew/Petteril areas. It is likely that different forms of erosion exert different influences on riparian and in-stream conditions and ultimately upon salmonid habitat. For example, fluvial activity and channel migration have been associated with woody debris abundance (Piegay *et al.*, 2000); and the availability of undercut bank cover (Belsky *et al.*, 1999). On the other hand, stock trampling has been associated with channel widening (Trimble and Mendel, 1995); bank slumping (Belsky *et al.*, 1999) and the loss of riffle-pool sequences (Gilvear *et al.*, 2002). These results suggest that as the scale of investigation contracts there is a reduction in the spatial variability of dominant erosive processes. As such, the primary causes of severe bank erosion and its impact upon in-stream conditions and salmonid habitat may become more identifiable. However, whilst spatial variability in the cause of erosion may decline, spatial variability in the presence/absence of severe erosion remains high at the area-scale (Figure 6.13(b)).

6.3.2.2 Regression analysis

To examine whether differences in the spatial variability of habitat controls at different scales of investigation result in different relationships between habitat and salmonid fry abundance, regression analysis was repeated at an area-scale using the 2004 salmonid fry data (Tables 6.11 and 6.12). Regression analysis was performed using the new set of transformed PCA factors created for each area, with the addition of the barrier variable in analysis of trout fry. Again salmon sites were only analysed below barriers and in all models probability limits for variable entry were set at 0.1, with variable removal set at 0.2. Due to restricted sample sizes at the area-scale, and variation in the level of salmonid fry abundance and distribution a combination of multiple forward stepwise regression and binary logistic regression (using forward likelihood ratio selection) was applied. Dependent variables also varied between areas dependent upon data normality as tested using Kolmogorov-Smirnov tests and class frequency distributions. Dependent variables included total salmon/trout ($n5min^{-1}$), Square root of total salmon/trout (SQRT), log of total salmon/trout plus 1 (Ln + 1), salmon/trout class and salmon/trout presence/absence. To aid in interpretation of the tables, PCA factor numbers have been followed by a description of the most significant variables represented by that factor.

In comparison with the catchment-scale analysis the results of the area-scale analysis indicate that different habitat controls are significant in explaining salmon and trout fry abundance/presence at different scales of investigation and in different locations of the catchment. Particularly notable was the increase in the number of in-stream and riparian-scale habitat controls that were found to be significant. Stock access, tunnelled vegetation, overhead tree cover, severe bank erosion, stock trampling, fluvial erosion and gravel presence were all found to varying extents to explain spatial variation in both salmon and trout fry populations at the area-scale. However, different controls were important in different areas.

Table 6.11: *Multiple regression models summarising relationships between habitat controls and salmon fry abundance or presence/absence in 2004 at the area-scale. Model fit is either the Adjusted R^2 in the case of linear regression or the Nagelkerke R^2 in the case of Binary logistic regression.*

Area	Dependent	Regression model	Regression equation	Model fit	Sample size
Tyne Gap	SQRT	Linear	= 3.006 + 1.321(Factor 4 Gravel dominant) – 0.494 (Factor 6 Stock access)	0.387	17
Pennine Becks	SQRT	Linear	= 3.687 – 1.068 (Factor 1 Increasing connectivity risk) – 0.697 (Factor 5 Tunnelled vegetation) – 0.599 (Factor 4 Increasing channel slope)	0.295	33
Orton & Howgill Fells			No significant variables were identified		37
Ullswater & Lowther Valley	Salmon class	Linear	= 1.421 + 0.856 (Factor 1 Increasing connectivity risk) + 0.459 (Factor 3 Increasing overhead cover)	0.496	48
Caldew & Petteril Rivers	Salmon presence	Binary logistic	= -19.021 – 77.006 (Factor 1 Increasing connectivity risk) + 32.633(Factor 6 Gravel dominant & no stock access)	1.000	27

Table 6.12: *Multiple regression models summarising relationships between habitat controls and trout fry abundance or presence/absence in 2004 at the area-scale.*

Area	Dependent	Regression model	Regression equation	Model fit	Sample size
Tyne Gap			No significant variables were identified		31
Pennine Becks	Trout class	Linear	= 2.243 + 0.757(Factor 3 Decreasing stock erosion & increasing fluvial erosion) – 0.844(Barrier)	0.262	46
Orton & Howgill Fells	Trout presence	Binary logistic	= 0.008 – 1.310(Factor 1 Increasing connectivity risk)	0.353	33
Ullswater & Lowther Valley	Trout class	Linear	= 1.795 – 0.686(Factor 1 Increasing connectivity risk) + 0.446(Factor 5 Gravel siltation present) + 0.430(Factor 6 Gravel dominant) -0.840(Barrier)	0.399	54
Caldew & Petteril Rivers	Trout presence	Binary logistic	= -1.207 – 1.867(Factor 4 Severe stock erosion) – 1.401(Factor 5 Fluvial erosion)	0.486	27

Catchment-scale habitat controls were still found to be important and catchment-scale land cover and hydrological connectivity were still the most significant control over spatial patterns of salmonid fry abundance and distribution in a number of areas, but not in all. Analysis at the catchment-scale suggested that salmon and trout fry exhibit a species-specific response to catchment land cover as filtered by hydrological connectivity. Salmon fry were related to non-linear catchment-channel connectivity risk whilst trout fry were related to linear risk. In contrast, at the area-scale, both species showed a similar linear, negative relationship with increasing catchment-channel hydrological connectivity risk. The only exception to this was salmon fry abundance in the Ullswater and Lowther Valley area where a linear, positive relationship was observed. This suggests that in the majority of areas it is high levels of land cover risk as filtered by catchment-channel connectivity that are limiting to salmonid populations (both salmon and trout) but that in the Ullswater and Lowther Valley area very low levels of connectivity may be limiting to salmon fry. This agrees with earlier discussion suggesting that it is only within this area that extreme low levels of connectivity are observed. It has been hypothesised that levels of catchment-channel connectivity are so low in this area that nutrient delivery and autochthonous production are reduced to such an extent that abundant juvenile salmon populations cannot be supported. Resistant volcanic geology in this area further promotes nutrient-poor soils and oligotrophic conditions. In this regard it is interesting to note that within the Ullswater and Lowther Valley area salmon fry abundance was positively related to increasing cover from riparian vegetation and trees. This may be related to increased food availability in such areas as a result of organic inputs from allochthonous production. In contrast, salmon fry abundance was negatively related to tunnelled riparian tree cover in the Pennine Beck area (Figure 6.16). These different responses may reflect area-scale structuring of salmon fry abundance in connection with the availability and abundance of various food sources, as influenced by the interaction of both catchment-scale and riparian-scale habitat controls. For example, in areas of optimal catchment-channel connectivity (e.g. Pennine Becks) invertebrate abundance is likely to be higher in open reaches associated with autochthonous production as opposed to shaded and particularly tunnelled reaches. Research has shown invertebrate and salmonid fry abundance to be reduced in extensive reaches (>400m in length) of tunnelled vegetation when compared with open or partly shaded reaches (O'Grady, 1993). However, in oligotrophic reaches where autochthonous production is restricted invertebrate abundance may actually be higher in shaded reaches where there are greater inputs of terrestrial invertebrates. This highlights the importance of considering interactions between habitat controls at different scales.

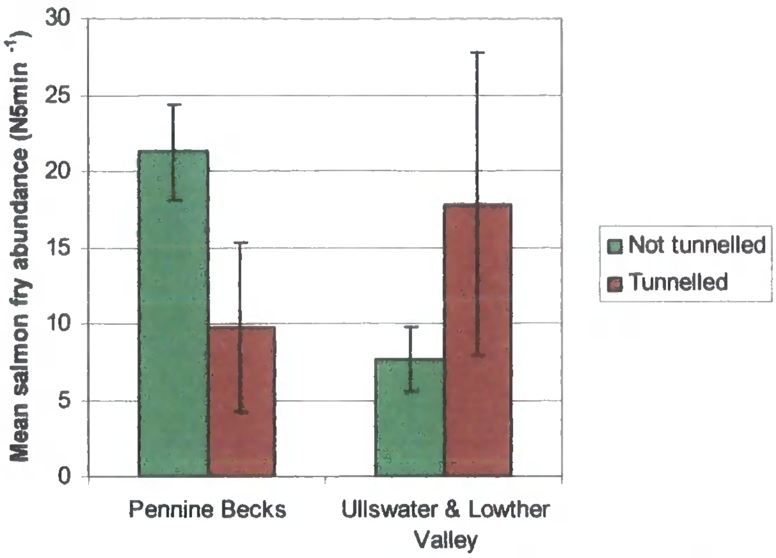


Figure 6.16: *Area-scale variations in mean salmon fry abundance stratified by the presence/absence of tunnelled riparian vegetation. Standard error bars are shown.*

A positive relationship between fry abundance/presence and the presence/dominance of gravel substrate was observed for salmon fry in the Tyne Gap and Caldew/Petteril areas and for trout fry in the Ullswater/Lowther Valley area. Bedrock outcrops and/or boulder substrates are typically dominant within these areas of the catchment and it may be that the availability of suitable spawning habitat is limiting recruitment to fry in these areas. Stock access and the presence of erosion due to stock were associated with a reduction in fry abundance/presence for salmon in the Tyne Gap and Caldew/Petteril areas and for trout in the Pennine Becks and Caldew/Petteril areas. This corresponds with current understanding of habitat controls suggesting that intensive grazing and stock access within the riparian zone can have a negative impact upon juvenile salmonid habitat through loss of cover, channel widening, increased width:depth ratios, elevated inputs of fine sediment, reduced terrestrial inputs and a reduction in riparian buffering functions (Chapter Two, Section 2.7.2). These variables were not found to be significant at a catchment-scale (with the exception of salmon presence/absence in 2005), a finding which may be related to their more localised extent resulting in them only exerting an identifiable influence over salmonid populations at smaller spatial scales. With regard to trout fry, the presence of impassable barriers was again associated with a lower abundance of fry in the Pennine Beck and Ullswater/Lowther Valley areas. However, it is interesting that overhead tree cover was no longer found to be significant in explaining spatial patterns of trout fry abundance at the area-scale. Finally, there was little relationship observed between fry abundance and the location of habitat occupied (as measured by channel slope) at the area-scale. The only exception to this was salmon fry abundance in the Pennine Becks area which was negatively related to increasing channel slope, indicating a preference for lower gradient channels.

6.3.2.3 Summary of area-scale analysis

Spatial variation in habitat controls was found to vary with the scale of investigation, particularly with regard to catchment-scale land cover as filtered by surface hydrological connectivity, for which variance decreased as the scale of investigation contracted. The results of PCA analysis showed that different relationships were observed between individual habitat controls at the area-scale compared with the catchment-scale, most notably for catchment-scale land cover and connectivity, gravel siltation and bank erosion. Differences were also reported in the relationships identified between salmonid abundance/distribution and habitat controls both between different scales of investigation and between different locations at the area-scale. This suggests that the scale and location of investigation is important in determining the relationships that will be identified between salmonid populations and habitat controls. An increase in the number of in-stream and riparian-scale controls explaining spatial patterns of abundance was observed at the area-scale compared with the catchment-scale. However, the significance and type of response of salmonid populations to these variables varied between areas and appeared in a number of cases (e.g. tunnelled vegetation) to still be determined by variations in catchment-scale controls between areas (e.g. land cover and hydrological connectivity). These results will be discussed in Chapter Seven in terms of Hypothesis (3) and the importance of considering investigation scale when undertaking research into and management of salmonid fisheries.

6.3.3 Relationships between habitat controls and salmonid parr

To enable testing of Hypothesis (1) and examine the extent to which relationships between habitat and salmonid abundance vary with life-stage, multivariate analysis was also applied to salmonid parr. As for fry data, sample sites were selected for inclusion in the data analysis according to the availability of all habitat variables at the site. 44 trout and 35 salmon parr sites surveyed in 2005 were therefore available for the analysis. Sites were distributed within seven tributaries of the Upper Eden catchment and had been selected for electrofishing to cover the range of habitat variables considered.

6.3.3.1 Principal components analysis

As presented in Table 6.1, a slightly different suite of habitat and fisheries variables were used in the analysis of parr data. To take account of this and the restricted area of application, PCA was repeated using the habitat data for the 44 selected sites. PCA was applied with a varimax rotation and 5 factors with an eigenvalue greater than 1 were extracted accounting for 73.1% of the variability in the original dataset (Table 6.13). All habitat variables selected for parr analysis (Table 6.1) were included with the exception of the barrier variable. Table 6.14 presents a summary of the habitat variables significantly represented by each of the factors and their relationship with those factors.

Table 6.13: *Rotated component matrix of habitat factors extracted for parr analysis with an eigenvalue >1. The factor to which each habitat control is most strongly related is highlighted in bold.*

Habitat control	Factor				
	1	2	3	4	5
In-stream habitat controls					
Dominant substrate	-.333	.275	-.270	-.352	**.364**
% pool habitat	.330	.292	**-.518**	.447	-.104
Channel width	-.070	.059	**.825**	.180	-.183
Channel slope	-.108	.080	-.062	**-.591**	-.225
Riparian-scale habitat controls					
Erosion presence	-.161	**.601**	-.362	-.106	.330
Stock access	-.202	**.666**	-.211	.023	.315
Erosion on both banks	.014	**.892**	-.010	.272	-.015
Erosion severity	.101	**.900**	.136	.093	-.017
Stock erosion	.136	**.800**	.232	-.307	-.040
Fluvial erosion	-.234	.120	-.035	**.824**	-.061
% overhead tree cover	.375	.293	.356	.236	**.573**
Tunnelled vegetation	.131	-.049	-.029	-.080	**-.824**
Riparian land use	.396	**.593**	-.026	-.047	.074
Catchment-scale habitat controls					
Catchment-channel hydrological connectivity risk (unweighted)					
Linear	**.958**	.041	.129	-.026	-.024
Classified linear	**.889**	.107	.116	-.163	-.147
Non-linear	.465	-.011	**.651**	-.139	.148
Catchment-channel hydrological connectivity risk weighted by land cover					
Linear	**.948**	.057	.051	.106	.032
Classified linear	**.930**	-.016	.061	.071	-.040
Non-linear	.397	.061	**.701**	-.043	.289

Table 6.14: *Summary of parr habitat variables represented by factors extracted using Principal Components Analysis*

Factor	Name	Correlation	Habitat variables represented
1	Linear connectivity	Positive	Linear catchment-channel hydrological connectivity (weighted and unweighted by land cover)
2	Bank erosion severity	Positive	Bank erosion, severe erosion, erosion on both banks, stock access, stock erosion and riparian land use
3	Channel width	Positive	Channel width & Non-linear catchment-channel hydrological connectivity (e.g. greater probability that risk falls outside the optimal range, Chapter 5, Section 5.4)
		Negative	Percentage pool habitat
4	Fluvial erosion	Positive	Fluvial erosion
		Negative	Channel slope
5	Overhead cover	Negative	Tunnelled vegetation
		Positive	Decreasing overhead tree cover & dominant substrate suitability

6.3.3.2 Correlation analysis

As for fry, correlation analysis was used to evaluate relationships between individual habitat controls and the abundance and presence/absence of salmonid parr. Kolmogorov-Smirnov tests again revealed the salmonid data to be non-normal even after transformation and a non-parametric Spearman Rank correlation was therefore applied (Table 6.15). In comparison with the salmonid fry data, few significant correlations with habitat controls were observed. Trout parr abundance and presence/absence were significantly and negatively correlated with decreasing overhead tree cover, linear and non-linear catchment-channel hydrological connectivity risk and linear and classified linear catchment-channel hydrological connectivity risk weighted by land cover. In addition, trout parr abundance was also negatively correlated with channel width. This suggests trout parr have a preference for narrow channels with a plentiful supply of overhead cover in areas of low to moderate catchment-channel hydrological connectivity risk. Unlike trout fry, no significant correlation with channel slope and erosion presence was observed. Salmon parr exhibited even fewer significant relationships with the habitat controls. Only a negative relationship between the presence of impassable barriers, and a positive relationship with the presence of tunnelled vegetation were observed for salmon parr abundance and presence/absence. In addition salmon parr abundance was also negatively correlated with the decreasing overhead tree cover. This unsurprisingly confirms that salmon parr are not found above impassable barriers and more unexpectedly indicates that salmon parr, like trout, have a preference for areas with a plentiful supply of overhead cover. This may suggest a change in behaviour for salmon from fry to parr, as although not significant salmon fry showed a negative relationship with increasing overhead cover.

Table 6.15: *Correlation analysis between individual habitat controls and salmonid parr abundance and presence/absence. Spearman rank correlation was applied and all tests were two-tailed. Significant correlations are shown in bold (*) p <0.05 and (**) p<0.001.*

Habitat control	Trout parr abundance (N100m^{-2})	Trout parr presence or absence	Salmon parr abundance (N100m^{-2})	Salmon parr presence or absence
In-stream habitat controls				
Dominant substrate	0.136	0.129	0.044	-.078
% pool habitat	-0.086	-0.057	-0.261	-.227
Channel width	**-0.326(*)**	-0.222	0.152	.222
Impassable barrier	-0.043	-0.166	**-0.486(**)**	**-.531(**)**
Channel slope	0.074	-0.037	0.084	-.066
Riparian-scale habitat controls				
Erosion presence	0.232	0.162	-0.025	-.086
Stock access	-0.060	0.020	-0.028	-.089
Erosion on both banks	-0.042	0.027	-0.092	-.101
Erosion severity	0.079	0.125	0.012	.029
Stock erosion	-0.135	-0.139	0.025	.006
Fluvial erosion	0.101	0.205	0.031	.033
% overhead tree cover	**-0.355(*)**	**-0.354(*)**	**-0.325(*)**	-.231
Tunnelled vegetation	0.119	0.257	**0.298(*)**	**.380(*)**
Riparian land use	-0.175	0.014	0.028	.100
Catchment-scale habitat controls				
Catchment-channel hydrological connectivity risk				
Linear	**-0.451(**)**	**-0.324(*)**	-0.228	.007
Classified linear	-0.283	-0.173	-0.134	.055
Non-linear	**-0.365(*)**	**-0.447(**)**	-0.102	.049
Catchment-channel hydrological connectivity risk weighted by land cover				
Linear	**-0.454(**)**	**-0.300(*)**	-0.240	-.014
Classified linear	**-0.458(**)**	**-0.324(*)**	-0.186	.060
Non-linear	-0.256	-0.275	-0.037	.058

One explanation for this apparent change in behaviour may be that, as juvenile salmon grow, substrate and turbulent flows may no longer provide adequate cover from predation and competition, with parr instead relying on overhead and bankside cover. Such forms of cover are also considered an important component of over-wintering habitat, which is increasingly being raised within the scientific literature as a critical limiting factor to the survival of salmonid parr

(Valdimarsson and Metcalfe, 1998; Armstrong and Griffiths, 2001; Armstrong *et al.*, 2003; Riley *et al.*, 2006). Unlike salmon fry, no relationships with channel width, channel slope or catchment-channel hydrological connectivity were observed. Fewer differences between salmon and trout were observed for parr than for fry, indicating that during the parr life-stage both salmon and trout may occupy and compete for similar habitats. This was supported by positive Spearman Rank correlations between trout parr and salmon parr abundance ($p < 0.01$) and between trout parr and salmon parr presence/absence ($p < 0.05$).

6.3.3.3 Regression analysis

Multiple regression analysis using the parr PCA factors was undertaken to establish whether habitat controls explain a significant proportion of the variation in salmonid parr populations and, if so, which variables are most significant. Due to the non-normal distribution of salmonid parr regression analysis was performed using ordinal dependents. Unlike fry, no standard classification system was available. Instead parr were grouped into five classes of zero fish plus four further classes of equal membership (Table 6.16).

Table 6.16: *Classification system applied to salmonid parr based on equal membership*

Class	N 100m^2	
	Trout parr	Salmon parr
A (Excellent)	>3.9	>6.8
B (Good)	2.9-3.9	4.0-6.8
C (Fair)	1.4-2.8	1.7-3.9
D (Poor)	0.1-1.3	0.1-1.6
E (Absent)	0	0

The 5 factors identified by PCA were then related to the ordinal classed fry data using forward stepwise multiple regression (Table 6.17). Probability limits for variable entry into the model were set at 0.1, with variable removal set at 0.2. The barrier variable was again added to the trout analysis as an independent variable as it had not been included in the PCA analysis. Additionally, the species-specific fry abundance within 2000m variables were included to account for any spatial variation in parr populations as a result of recruitment. T-statistics and *p* values are again presented for each significant variable to aid interpretation (Table 6.18).

Table 6.17: *Multiple regression models summarising relationships between habitat variables and salmonid parr for an area of the Upper Eden catchment.*

Dependent	Regression equation	Adjusted R^2	Sample size
Salmon parr	= 1.728 + 0.064(Salmon fry abundance) – 0.736 (Factor 1)	75.8	35
Trout parr	= 2.471 + 0.058(Trout fry abundance) – 0.397 (Factor 1)	18.4	44

Table 6.18: *T-statistics and p values for parr multiple regression models*

Model	t statistic	*p* value
Salmon parr		
Salmon fry abundance	7.649	0.000
Factor 1	-5.040	0.000
Trout parr		
Trout fry abundance	2.101	0.042
Factor 1	-1.807	0.078

Adjusted R^2 values indicate that the fisheries and habitat variables do explain a significant proportion of the spatial variation in salmon and trout parr. Particularly striking is the very high adjusted R^2 value of 75.8% reported for salmon parr. Of this 58% was attributed to the level of salmon fry abundance within 2,000 metres of the parr electrofishing site. A high t-statistic (7.649) and highly significant *p*-value (<0.00001) suggest that the level of salmon fry productivity is extremely important in determining salmon parr abundance within the area of the catchment studied. Greater numbers of salmon parr were found within 2,000 metres of high fry abundance. Trout parr abundance was also observed to be positively related to fry productivity although to a considerably lesser degree than salmon. An R^2 value of only 18.4% was reported, of which 14% related to fry abundance, the t-statistic was not as strong (2.101) and the *p*-value was less significant (<0.05). The three hypotheses for the lower explanation of spatial variance in trout compared with salmon proposed in Section 6.3.1.4 may also be relevant here. These results suggest that it may be fry production rather than parr habitat which determines parr abundance in this area of the catchment. This finding has important implications for targeting fisheries restoration and will be discussed further in Chapter Seven. However, a significant proportion of spatial variation in both salmon (17.8%) and trout (4.4%) parr abundance was also explained by a negative relationship with Factor 1 representing lower abundance at increased levels of catchment-channel connectivity risk. This further emphasises the critical role of catchment-scale land cover as filtered by surface and shallow sub-surface hydrological connectivity in structuring salmonid populations, in this case at the parr life-stage. It is interesting to note that for both species it is the linear connectivity variables that are significant in explaining spatial variation indicating that it is high levels of connectivity which are limiting to parr of both species, at least in this area of the catchment. This contrasts with the findings for salmonid fry which suggested that salmon fry were related to the non-linear connectivity variable, also limited by low levels of connectivity. Detailed examination of the dataset suggests that this may be an artefact of the data applied. The area of the catchment studied here does not experience the very low levels of connectivity risk found to be limiting to salmon fry production in other areas of the catchment (e.g. Ullswater and Lowther Valley). Further analysis in these areas would be required to ascertain if low levels of catchment-channel connectivity risk are limiting to parr. Land cover and connectivity impacts on water quality are considered density-independent habitat controls (e.g. Milner *et al.*, 2003) and it should be noted that unlike fry, particularly trout fry, no density-dependent factors (e.g. overhead cover, bank erosion, substrate) were found to significantly explain spatial variation in parr abundance. It has been suggested that this may be related to the greater dispersal capabilities of parr, which can escape density-dependent controls, a hypothesis that is discussed further in Chapter Seven. In contrast to the fry analysis, both salmon and trout parr abundance appear controlled by very similar variables with no significant distinction in the scale or location of habitat occupied. This is supported by the results of correlation analysis which found the presence of tout parr to correspond positively with the presence of salmon parr indicating that both species are utilising similar habitats. BLR was also applied to the parr data to assess whether there were any differences between salmonid distribution and habitat compared with abundance (Table 6.19).

Table 6.19: *Binary logistic regression models summarising relationships between habitat controls and salmonid parr presence/absence for an area of the Upper Eden catchment. A forward likelihood ratio selection of variables was applied. Probability limits for variable entry into the model were set at 0.1, with variable removal set at 0.2.*

Dependent	Regression equation	Nagelkerke R^2	Sample size
Salmon fry	= -22.175 + 2.3(Salmon fry abundance) + 4.946(Factor 5 Overhead cover)	0.955	35
Trout fry	=1.784 – 1.064(Factor 1 Linear connectivity) – 1.983 (Barrier) – 0.990 (Factor 3 Channel width) – 0.796 (Factor 5 Overhead cover)	0.408	44

Similar to abundance data, the presence of salmon parr was positively related to salmon fry abundance with again an exceptionally high goodness of fit statistic reported (Nagelkerke R^2 = 0.955). A positive relationship with Factor 5 (decreasing overhead cover) was observed contrasting with results of the correlation analysis where a negative relationship was observed.

Factor 5 entered the model with a probability score of 0.093 and only increased performance slightly with an increase in the Nagelkerke R^2 from 0.912 to 0.955. As such the reliability of this is result is uncertain, particularly in light of the contrasting correlation results. The BLR model for trout presence/absence exhibited a lower goodness of fit (Nagelkerke R^2 = 0.408) with trout presence negatively related to Factors 1, 3, 5 and the barrier variable. This suggests that there is a higher probability of trout parr presence in streams of low catchment-channel hydrological connectivity risk which are narrow, are below impassable barriers and have a greater proportion of pool habitat and large amounts of overhead cover. In contrast to the abundance data, no relationship with trout fry was observed. This is an interesting result as it suggests that, whilst habitat controls the distribution of trout parr, it is recruitment that influences abundance. In terms of management this raises the possibility that restoring parr habitat may simply result in a redistribution of fish rather than an increase in overall abundance unless recruitment from fry also increases. It was not possible to undertake analysis at the area-scale due to the limited availability of data for parr.

6.3.3.4 Summary of salmonid parr analysis

Spatial patterns in the abundance of salmonid parr were found to be strongly influenced by the abundance and productivity of salmonid fry within 2,000m of the parr's territory, more so than by any of the habitat variables. Catchment-scale land cover and hydrological connectivity risk factors were the only habitat controls found to explain significant spatial variation in salmon and trout parr abundance. In terms of distribution, salmon parr were again significantly related to fry abundance. However, trout parr distribution was explained by a combination of habitat factors, suggesting that different controls may be responsible for determining trout parr distribution compared with abundance. Finally, a positive correlation between the presence and abundance of trout parr and the presence and abundance of salmon parr suggested that they may be in competition for the same habitats at this life-stage. These results are to be discussed in Chapter Seven in relation to Hypothesis (1) and the level of mobility and potential for dispersal of salmonid parr compared with salmonid fry.

6.4 Chapter Summary

This aim of this chapter was to analyse relationships between habitat controls and salmonid abundance/distribution presenting the results required to achieve Objective (3) of this thesis: the testing of hypotheses proposed in Chapter Two, and discussion of results in the context of approaches to prioritising habitat restoration. This was achieved in two stages. First, a spatially-structured hierarchical GIS was developed integrating the habitat and salmonid population data acquired in Chapters Three, Four and Five. This involved the spatial co-registration of data and coding of variables for analysis. Second, an approach capable of interrogating large multivariate data sets was identified and applied. Correlation and PCA analysis were used to examine relationships between habitat controls and to assess and to address the issue of variable collinearity. The results suggested that in specific locations of the landscape and at specific scales of habitat, specific combinations of habitat pressures are likely to occur. The results also highlighted the significant influence that land cover can have resulting in multiple pressures on the freshwater environment at multiple scales. This was then followed by regression analysis to examine relationships between salmonid abundance (forward step-wise multiple regression) and salmonid presence/absence (binary logistic regression). Analysis was undertaken in three stages. First, habitat controls were related to salmon and trout fry data at the catchment-scale. Results were presented showing that habitat controls do explain an acceptable proportion of the spatial variation observed in salmonid abundance/distribution. However, substantial species-specific differences were observed in the relationships identified which will be discussed in Chapter Seven in relation to Hypothesis (2) and the scale of habitat occupied by each species. Second, analysis considered the impact of investigation scale upon habitat controls and their relationship with each other and salmonid populations. Unfortunately, restrictions in the number of parr population sample sites meant that analysis could only be applied at an area-scale and not at a tributary-scale as had been hoped. Different relationships between habitat controls and between habitat and salmonid abundance/distribution were observed at different scales of investigation and between different locations at the area-scale. This highlighted the importance of considering scale in research such as this and will be discussed in Chapter Seven with regards to Hypothesis (3) and implications for research into and the management of fisheries, particularly salmonid fisheries. Finally, analysis was undertaken relating habitat controls to salmonid parr abundance for an area of the Upper Eden catchment. Results were presented suggesting that parr abundance may be more related to fry productivity and recruitment than to habitat with the exception of catchment-scale hydrological connectivity. These findings will be discussed in Chapter Seven in relation to Hypothesis (1) and differences in the level of mobility and dispersion at the fry and parr life-stage. Chapter Seven will now discuss in more detail the results presented in this chapter in relation to the three hypotheses proposed in Chapter Two. This will be followed by a discussion of the implications of these findings for the development of effective approaches to fisheries management and habitat restoration using the Eden catchment as a case study.

Chapter Seven - Relationships between habitat and salmonids: the importance of scale and effective approaches to fisheries management.

7.1 Introduction

Chapter Six presented the results of multivariate analysis which evaluated the relationship between habitat controls and salmonid populations within the Eden catchment. Analysis was undertaken at a range of spatial scales stratified by both life-stage and species. The aim of this chapter is to integrate these results in relation to: (1) the three hypotheses proposed in Chapter Two; and (2) approaches to fisheries management. Throughout this research, and in particular, throughout this discussion, it is vital to recognise that although multivariate analysis indicates significant relationships between variables, this cannot guarantee that causal explanation of spatial variance is identified correctly. Instead, the relationships identified represent association not causation (Wiley *et al.*, 1997). The identification of no association may be indicative of no causation. The identification of association needs other supporting arguments for linkages to be causal. Thus, this discussion supports the associations identified empirically with a broader and deeper investigation in relation to other supporting evidence.

7.2 Hypothesis (1): *Relationships between habitat and salmonid abundance/distribution are structured by life-stage according to the level of mobility and potential for dispersal at each life-stage.*

The impact of habitat at various stages of the salmonid life-cycle has been assumed to be determined by a fish's mobility and capability for dispersal at a given life-stage (Milner *et al.*, 2003). In particular, it has been suggested that the less mobile a fish, the more likely it is to be sensitive to localised habitat pressures and regulated by density-dependent mechanisms. As mobility increases, it is argued that fish stocks can disperse, distributing their population over a wider area to take advantage of spatially dispersed habitat suitable to their life-stage (Armstrong, 2005). As such, mobile fish may be more capable of avoiding localised pressures and reducing susceptibility to density-dependent regulation mechanisms. Instead, it is hypothesised that more mobile fish will be regulated by density-independent pressures (Milner *et al.*, 2003). In terms of salmonids, research has shown fry to be the least mobile life-stage with little increase in overall dispersal range as density increases (e.g. Armstrong, 2005; Einum and Nislow, 2005). It is therefore assumed that habitat bottlenecks (Section 2.4.2) and density-dependent mortality occur most frequently at this life-stage (Nislow *et al.*, 2004).

However, habitat bottlenecks have been observed at older life-stages (Elliot and Hurley, 1998). This may occur if habitat pressures are extensively distributed throughout the spatial range of mobility at that life-stage. For example, a lack of over-wintering habitat (Armstrong and Griffiths, 2001) or pool habitat (Rincon and Lobon-Cervia, 2002) may cause density-dependent mortality of older fish. Alternatively, in environments where spawning and fry habitat is abundant and spatially distributed, recruitment is likely to be high and the carrying capacity of the environment for older life-stages may become saturated, stimulating density-dependent regulation and self-thinning as larger fish require more food and larger territories (Armstrong, 2005). In terms of management, it is important to determine the life-stage at which habitat bottlenecks are occurring as it is by improving habitat relevant to this life-stage that the greatest improvements in stocks are likely to be achieved. The example was given in Chapter One (Section 1.2.2) of ineffective restoration where fry habitat was improved in an area where parr are limited by a lack of over-wintering habitat (Armstrong *et al.*, 2003). In such an area, the most effective restoration approach would be to improve parr habitat. Different life-stages also have different habitat requirements and the type of habitat restoration required may therefore vary, dependent upon the life-stage that is limited.

7.2.1 Discussion of Eden catchment results in the context of Hypothesis (1)

To examine whether relationships between habitat and salmonid abundance are structured by life-stage within the Eden catchment and determine which life-stages, if any, are limited by habitat, both fry ($n5min^{-1}$) and parr ($n100m^{-2}$) abundance were related to habitat controls using multivariate statistics. It should be remembered that the robustness of the findings presented are limited by the availability of fisheries data. Parr data were only available for 2005 and for a limited number of sites in comparison with the catchment-wide data available for fry. They have been compared to the results of the 2004 fry analysis. Hypothesis (1) has been broken down into two parts for discussion. First, were different relationships found between habitat and salmonid abundance for fry compared with parr? Second, if any differences were observed, is there any evidence suggesting that this is related to differences in the level of mobility and potential for dispersal at each life-stage? In respect of the first question differences were found in relationships between habitat and salmonid abundance at each life-stage as assessed through regression analysis (Table 7.1).

Table 7.1: *Relationships between habitat controls and salmonid abundance stratified by life-stage. Variables presented are those selected as significant by forward stepwise regression analysis.*

Life-stage	Salmon	Cumulative adjusted R^2	Trout	Cumulative adjusted R^2
Parr	Fry abundance	57.9%	Fry abundance	14.0%
	Land cover and catchment-channel connectivity	75.8%	Land cover and catchment-channel connectivity	18.4%
Fry	Land cover and catchment-channel connectivity	26.8%	Percentage overhead cover & tunnelled vegetation	8.9%
	Channel slope & biotope	32.2%	Land cover and catchment-channel connectivity	15.4%
			Channel slope & biotope	20.1%
			Impassable barriers	24.6%

In particular, habitat controls were found to explain less of the spatial variation in abundance for parr than fry. Similar findings were reported for correlation analysis between individual habitat controls and salmonid abundance, with fewer significant correlations observed for parr than for fry (Section 6.3.1.3). It has been proposed that habitat has its strongest effects when the standing stock approaches the carrying capacity of the environment (Armstrong *et al.*, 2003). Therefore, diminished relationships between habitat and parr abundance may suggest that parr are not approaching the existing carrying capacity of the environment. In addition, a highly significant relationship between parr abundance and fry abundance within a 2000m radius suggests that parr abundance may be more dependent upon fry productivity and recruitment than upon habitat. Similar observations have been reported for other systems. For example, fry density in Shelligan Burn, explained 66% of the spatial variance in parr (Gardiner, unpublished, *cited in* Milner *et al.*, 2003). These findings support research promoting the concept of a critical period for survival following emergence when strong density-dependent mortality may regulate overall population, reducing competition and the potential for density-dependent mortality at older life-stages (Nislow *et al.*, 2004). If habitat was limiting parr abundance, such a strong relationship with fry abundance would be less likely as high fry numbers would be constrained at the parr life-stage by habitat controls, resulting in the same number of parr as in areas of low fry abundance (Figure 7.1).

Figure 7.1: *Hypothetical representation comparing recruitment in a parr habitat limited system and a recruitment from fry limited system.*

Interestingly, the only habitat control to explain a significant proportion of the spatial variation in parr abundance was catchment-scale land cover, as filtered by catchment-channel surface hydrological connectivity risk, with high levels of risk corresponding with lower parr abundance. As discussed in Chapter Five (Section 5.4), land cover and hydrological connectivity are considered to potentially impact salmonid abundance through: (1) the delivery of fine sediment and siltation of spawning gravels; and (2) the delivery of solutes and impacts on water quality relevant to all life-stages. The impacts of pollution and water quality are considered within the scientific literature to represent density-independent mortality of salmonids (Milner *et al.*, 2003). In other words, poor water quality will act to proportionately limit salmonid parr abundance regardless of density and the level of recruitment from fry either through direct toxicity or chronic physiological impacts on health (e.g. Lower and Moore, 2003; Waring and Moore, 2004). These findings support research suggesting that parr populations are more likely to be regulated by density-independent mechanisms than density-dependent ones (Milner *et al.*, 2003). Over time, the impact of density-independent events is considered to reduce the amount of spatial variation explained by fry survival. For example, initial egg density explained 95% of the spatial variability in sea trout fry within Black Brows Beck, Cumbria, but only 44% of the variance in returning spawners (Elliot, 1994). Further investigation studying older life-stages would be required to ascertain if this is the case for the Eden catchment.

These findings do not mean that salmonid parr do not have any physical habitat preferences, just that they are currently not significantly limiting abundance, relative to fry productivity and the degree of catchment-channel hydrological connectivity. Binary logistic regression (BLR) suggested that a significant proportion of trout parr distribution (presence/absence) was explained by habitat including hydrological connectivity and land cover, overhead tree cover, percentage of pool habitat, channel width and impassable barriers. Correlation analysis also revealed that both salmon and trout parr abundances were positively correlated with increasing amounts of cover available from riparian vegetation and trees. This agrees with personal observations made in the field during electrofishing surveys where parr were commonly found in shaded areas amongst exposed tree roots, woody debris and below undercut banks with overhanging vegetation. In the absence of all natural forms of cover, one trout parr was even observed sheltering below a sheet of corrugated metal lodged in the channel bed. This also corresponds with research using PIT tags to monitor salmonid micro-habitat selection which highlighted the importance of cover for salmonid parr, both salmon and trout (Riley *et al.*, 2006). Thus, if fry productivity is increased and water quality improved, parr abundance would eventually become limited by the amount of available cover within the Eden catchment. Additionally, it was noted in Chapter Six (Section 6.3.3.2), that the positive correlation between cover and salmon parr appeared to represent a change in behaviour from the fry life-stage, that may be associated with a need for over-wintering habitat and more extensive cover at the parr life-stage (Armstrong and Griffiths, 2001). This potential change in behaviour highlights the importance of considering life-stage specific habitat requirements when developing restoration strategies as discussed in Chapter One (Section 1.2.2). For example, it may only be beneficial to undertake coppicing around salmon fry habitat (e.g. riffles). The same strategy may actually be detrimental if applied to salmon parr habitat (e.g. deeper, faster flowing reaches). Consideration of such factors allows for much more effective and economic restoration. A limited budget for coppicing could be targeted at a restricted habitat type where it will maximise benefits (e.g. salmon fry habitat) covering a much wider area of the catchment than if it was applied ineffectively to coppicing around all habitats in a smaller area of the catchment. Understanding life-stage specific habitat requirements should help develop precision restoration where certain strategies are restricted to specific habitat types.

In respect of the second question associated with Hypothesis (1), there does appear to be evidence supporting the hypothesis that relationships with habitat are structured by life-stage in relation to the potential for dispersal at each life stage. Salmonid fry abundance and presence were associated with channel width and channel slope, variables which exhibit distinct spatial structuring within catchments. Salmon fry were associated with wider channels of lower gradient, whilst trout fry were associated with narrow channels of steeper gradient. Trout fry presence was also negatively correlated to salmon fry presence suggesting that fry are restricted to distinct locations within the catchment dependent upon species-specific spawning requirements, showing little evidence of dispersal. This agrees with research suggesting that salmon and trout spawn in distinct areas and that the size of channel utilised for spawning is proportional to the size of the fish, salmon typically being larger than trout (Summers *et al.*, 1996). It is currently understood that species-specific selection of spawning habitat is related to variations in depth and velocity between channels of different scale. Wider channels are typically associated with deeper, faster flowing water than narrow channels. Hence, in wider channels, larger fish are more able to hold station whilst redd building (Crisp, 1996), but in narrow channels, shallow water depths restrict access and cover for larger fish enabling smaller fish to dominate (Armstrong *et al.*, 2003). These results correspond with the theory that fry emerge from redds and remain within discrete locations of the catchment where suitable spawning conditions are found. As such, a relatively small proportion of all stream habitat may provide suitable territories for fry, associated with distance from spawning habitat (Nislow *et al.*, 2004). It has been suggested that, although a river may be at maximum capacity for producing salmonids, in that all spawning sites are fully exploited, much of the habitat for juveniles may be under-utilised as they cannot disperse to occupy it (Armstrong, 2005) leading to the clustering of fry in specific locations. Interestingly, a positive correlation between salmon presence and the presence of gravels was also observed, again suggesting the dependence of salmon fry abundance upon proximity to spawning habitat (Hendry and Cragg-Hine 1997). Conversely, salmon parr exhibited no significant relationships with channel width or slope, whilst only trout parr presence showed a negative relationship with channel width. In addition, trout parr presence was positively related to salmon parr presence. These findings indicate that parr have a wider spatial distribution than fry, suggesting that they have dispersed to occupy a greater spatial extent and wider range of habitats throughout the catchment with the two species now co-habiting. This agrees with current scientific understanding presented in Chapters One (Section 1.2.2) and Two (Section 2.5.2), which suggests that parr are both able to disperse across greater spatial extents than fry, and are more capable of utilising a greater range of velocities and habitats than fry due to their improved swimming capabilities and dominance over smaller fish (Crisp, 1996).

An additional observation for trout parr in support of the hypothesis of habitat relationships structured by dispersal is that, whilst distribution (presence/absence) was explained by a number of density-dependent habitat variables (e.g. overhead cover and pool habitat), the same relationships were not apparent for trout parr abundance. In other words, at the parr life-stage different relationships with habitat are observed for presence compared with abundance data. It is suggested that the strong relationship between habitat and parr presence (distribution) reflects the ability of parr to structure their population so that they avoid localised reaches of poor habitat.

However, if overall abundance does not reach the carrying capacity of the environment, there may be no relationship between habitat and abundance as some areas of good habitat will be under-utilised leading to high variability in the dataset (Figure 7.2). Further, areas of excellent habitat may only be utilised by a few fish whilst areas of sub-optimal habitat contain more fish due to their relative position in relation to areas of fry productivity. As mentioned in Chapter Six, this raises the possibility that restoring parr habitat may simply result in a redistribution of fish rather than an increase in overall abundance, unless recruitment from fry also increases. At the trout fry life-stage, presence and abundance do not exhibit different relationships with habitat. Fry are less mobile and therefore less able to structure their population to avoid localised habitat pressures, instead dying in situ. As such, if fish are present (1000s of eggs laid nearby) their abundance is more likely to track habitat quality.

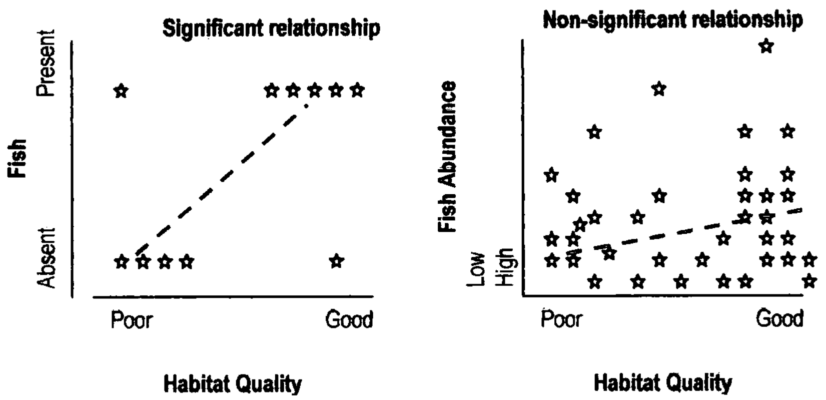


Figure 7.2: *Diagrammatic representation of differences in trout parr habitat relationships between (a) parr distribution and (b) parr abundance.*

Greater mobility provides parr with two advantages over fry. First, parr are more capable of dispersing until they find optimal habitat free from competition. As such, larger fish can use areas of stream habitat that could not be reached by fry during the critical period of density-dependent regulation due to distance from spawning habitat (Nislow *et al.*, 2004). Second, even if parr do not find optimal habitat within their range, they are more capable of adapting to the habitat available than fry which have more specific requirements. For example, Riley *et al.* (2006) observed trout and salmon parr to utilise a wide range of habitats from silty to coarse substrates, slow flowing pools to fast flowing reaches, areas of exposed tree roots to reaches with aquatic weeds. Subsequently, the amount of available resources may potentially increase as fish grow preventing self-thinning from occurring (Nislow *et al.*, 2004). Together, these two factors mean that parr are less likely to be limited by localised habitat pressures and density-dependent regulation than fry, justifying a focus on fry in habitat restoration. This ability may in part account for the dampened relationship between habitat constraints and abundance observed at the parr life-stage.

The only habitat variable to which parr abundance is significantly related is catchment-scale land cover, as filtered by hydrological connectivity. As discussed, water quality is commonly considered to impact salmonids through density-independent mechanisms. However, it may also exert density-dependent regulation through reductions in the availability of food and oxygen as a result of variations in nutrient delivery and biological oxygen demand. Results of the area-scale analysis (Section 6.3.2) suggested that this variable exerts its influence over a wider spatial extent (lower spatial variance) than other habitat variables such as bank erosion, cover and substrate that are more localised in extent (higher spatial variance) within the Eden catchment. It may therefore be the case that, where diffuse pollution occurs, poor water quality is distributed across a greater spatial extent than the spatial range of salmonid parr. For example, whole sub-catchments were predicted to exhibit land cover and connectivity risks within the highest risk category (based on equal class membership) (Figure 5.4). Consequently, parr may be unable to disperse over a great enough extent to avoid regulation. As such, catchment land cover and hydrological connectivity may trigger density-dependent mechanisms and constrain carrying capacity at much lower parr abundance than other more localised habitat pressures such as lack of cover. Based on the data available, it is not possible to discriminate between density-dependent and density-independent mechanisms in relation to the relationship observed between hydrological connectivity and parr abundance but, it is important to recognise that a single habitat factor may exert control over abundance in different ways. For example, if hydrological connectivity is thought to exert only density-independent regulation, stocking of parr may be proposed to improve stocks. If density-dependent mechanisms are also operating stocking may be of little benefit, as fish numbers may already be at the carrying capacity of the environment. Increasing populations beyond this is only likely to result in higher mortality or emigration (Aprahamian *et al.*, 2003).

7.2.2 Summary of Hypothesis (1) discussion

In summary, relationships between habitat and salmonid abundance within the Eden catchment were observed to be structured by life-stage reflecting the level of mobility and potential for dispersal at each life-stage. Salmonid parr were found to be dispersed across a wider spatial extent than fry utilising a wider range of habitats. At the same time, less of their spatial variation in abundance was explained by habitat variables. This suggests that due to their greater mobility and adaptability, parr are more able to avoid localised habitat pressures. As a result, parr abundance at a specific site is less regulated by localised pressures and carrying capacity, instead being determined by productivity at earlier life-stages. The exception to this is the impact of land cover upon water quality as filtered by hydrological connectivity. It is thought that this variable may act to limit parr abundance both through density-independent mechanisms such as toxicity in areas of both low and high recruitment and by density-dependent mechanisms such as competition for food and oxygen availability in areas of high recruitment due to its particularly extensive distribution and more endemic nature within the environment.

Based on these findings, it is proposed that the availability of parr habitat is not limiting abundance in the area of the Upper Eden catchment surveyed (Figure 6.1), and efforts to restore parr habitat here may be ineffective in improving stocks. Instead, it appears that the physical habitat bottleneck is occurring at the previous fry or spawning life-stage and it is suggested that restoring habitat at these life-stages will be more beneficial to improving stocks. To be effective, this restoration should be solely targeted at fry or spawning habitat, as life-stage specific habitat requirements may make intended restoration detrimental to other life-stages if applied to the wrong habitat. The type of restoration recommended, specific to each species considered, will be discussed in the following section in relation to Hypothesis (2). However, efforts to increase the environment's resilience to the delivery of diffuse pollution are likely to be beneficial in maximising survival of those fish which do survive to the parr life-stage. This may include strategies that may reduce the risk of catchment-channel hydrological connectivity (e.g. buffer strips) or changes to land management that minimise pollution generation in sensitive areas of the landscape (e.g. reduced fertiliser and slurry applications in highly connected fields). Further, should restoration be successful in improving fry survival, relationships between parr and habitat should be re-evaluated to assess whether subsequent habitat constraints develop at this life-stage.

7.3 Hypothesis (2): *Relationships between habitat and salmonid abundance/distribution are species and location specific relating to the scale of habitat occupied by different species.*

Research has proposed that, whilst juvenile Atlantic salmon and brown trout habitat requirements do overlap (Armstrong *et al.*, 2003), there are also a number of subtle differences between species, leading to the theory of niche separation as an important mechanism allowing the two species to co-exist (Riley *et al.*, 2006). Competition between the two species is widely acknowledged (Heggenes *et al.*, 1999), with juvenile trout considered to be more territorial and aggressive than salmon of the same age (Elliot, 1994). In particular, it is considered that juvenile trout will out-compete salmon in narrow streams where bankside cover is abundant relative to the area of the stream bed. The importance of overhead cover to young trout is well documented (Riley *et al.*, 2006) and they have been observed to defend such territories aggressively (Elliot, 1994). In wider streams, trout performance is thought to be reduced due to a reduction in the abundance of overhead cover relative to the stream bed. Here, salmon which are better adapted to hold station at high velocities (Crisp, 1996), are considered to dominate free from competition with trout (Armstrong *et al.*, 2003). Acknowledgement of species-specific requirements is important as individual restoration strategies may benefit different species to differing degrees. Further, whilst an individual strategy may be beneficial to one species, it may actually be detrimental to another. For example, increasing overhead cover in a reach currently dominated by juvenile salmon may increase trout densities leading to increased inter-species competition and ultimately a decline in juvenile salmon numbers (Armstrong *et al.*, 2003). Understanding these differences and interactions is important if effective restoration is to be delivered preserving habitat heterogeneity for all species. Testing the hypothesis that salmon and trout utilise different scales of habitat to different degrees is also important as different scales of habitat are found in different locations of the catchment and are subject to different habitat pressures as a result of both their varying location and scale. Consequently, the effectiveness of different restoration strategies may depend upon the scale and location of habitat it is applied to, relative to the dominant species found there.

7.3.1 Discussion of Eden catchment results in the context of Hypothesis (2)

To examine whether relationships between habitat controls and salmonids within the Eden catchment are species-specific and stratified by habitat scale, data for each species (Atlantic salmon and brown trout) were related to habitat controls individually throughout all analyses. Again, the hypothesis is broken down into separate parts for discussion. First, were different scales of habitat utilised to different degrees by salmon compared with trout? Second, were species-specific relationships between habitat and fish abundance/distribution observed and, if so, can these be related to the scale of habitat utilised? With respect to the first question, salmonid fry do demonstrate a clear species-specific response to the scale of habitat occupied. Figure 7.3 illustrates that trout fry abundance in 2004 was greatest in narrow streams (mean channel width ~ 6m) whilst salmon fry abundance was greatest in wider streams (mean channel width ~10m). As discussed in the previous section, this segregation is likely to reflect species-specific differences in spawning locations, associated with fish size, water depth and velocity. This corresponds with research presented in Chapter Two (Section 2.5.1) suggesting that trout are able to utilise much smaller streams than salmon for spawning (e.g. Elliot, 1994).

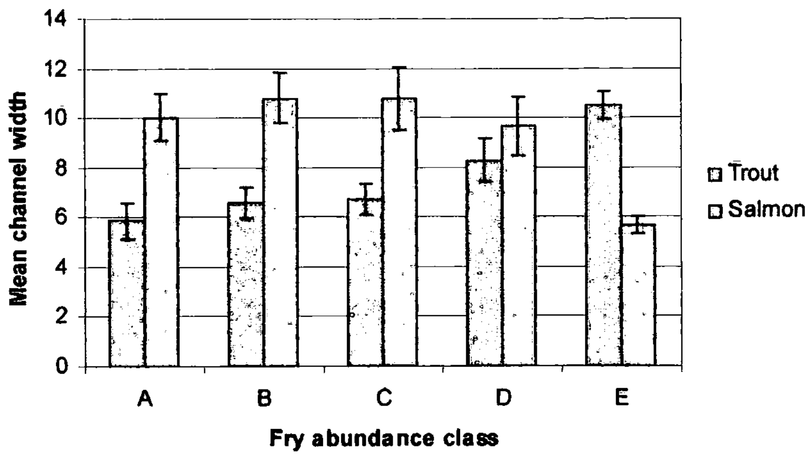


Figure 7.3: *Species-specific variations in the scale of habitat (measured by mean channel width) utilised by salmonid fry in 2004. Abundance classes refer to the classification system presented in Chapter 3, Table 3.5.*

With respect to the second question, species-specific relationships with habitat were observed at the fry life-stage. As presented in Tables 6.7 and 6.9, spatial variability in salmon fry abundance/distribution was most significantly explained by a non-linear relationship with hydrological connectivity risk weighted by land cover, and a preference for streams of lower gradient. Spatial variability in trout fry abundance/distribution was most significantly explained by the amount of overhead cover available, a linear relationship with hydrological connectivity risk weighted by land cover, the presence of impassable barriers and a preference for channels of steeper gradient. By considering the results of this research in the wider context of scientific literature, there is evidence to suggest that these species-specific relationships may be related to the scale of habitat occupied. First, habitats of different scales are typically found in different locations of the catchment. Wider streams are typically associated with lowland valley floodplains where channel gradients are lower, an assumption confirmed by Principal Components Analysis (PCA) (Table 6.3). This corresponds with the negative relationship reported by regression analysis between salmon fry and increasing channel gradient. This is also supported by research that suggests salmon spawn in channels with a slope less than 3% (Hendry and Cragg-Hine, 1997). Conversely, it is assumed and confirmed by PCA that, smaller streams dominate in upland environments where channel gradients are typically steeper. This corresponds with the positive relationship reported by regression analysis between trout fry and increasing channel gradient. These results emphasise the fact that different scales of habitat and different species are found in different locations of the catchment. Therefore, different species-specific restoration strategies may be required in different locations of the catchment. However, it should be remembered that narrow tributaries can also be found in lowland environments. In terms of salmonid parr, no relationships with channel gradient were observed for either species. As discussed in the previous section, this suggests that as fish grow and disperse they are able to utilise habitat in a wider range of locations throughout the catchment. (Riley *et al.*, 2006). Consequently, species-specific responses to habitat may become dampened.

Second, differences between salmon and trout fry response to land cover, as filtered by surface hydrological connectivity, were observed. Salmon fry exhibited a non-linear relationship with hydrological connectivity risk in that both low and high levels of risk resulted in reduced abundance. Trout fry exhibited a linear relationship in that only high levels of risk resulted in reduced abundance. This difference was discussed in Chapter Five (Section 5.4) and Chapter Six (Section 6.3.1) in the context of feeding habits. Juvenile salmon are considered particularly dependent upon autochthonous production (Summers, 2002), whilst terrestrial invertebrates from riparian vegetation and tree cover have been found to contribute significantly, up to 91%, to trout prey (Johansen *et al.*, 2005). It has been proposed that the different response to low levels of connectivity reflects the greater dependence of salmon fry upon autochthonous production which requires nutrient inputs to be delivered to the channel through higher connectivity with the wider landscape. Correlation analysis (Figures 6.4 & 6.5) suggested that lower levels of hydrological connectivity are associated with steeper channel slopes. PCA analysis also suggested that narrow streams are more prone to extremes of hydrological connectivity risk (Table 6.3). As such, narrow streams in upland environments utilised by trout fry, are more likely to represent marginal, low biomass environments, in terms of the availability of autochthonous food resources. This may explain the greater dependence of trout upon terrestrial sources of invertebrates associated with increased riparian cover compared with salmon. In contrast, high levels of catchment-channel hydrological connectivity and land cover risk were hypothesised to be detrimental to both species. This finding raises the issue that not all habitat controls may be related to the scale or location of habitat occupied. In the case of water quality, degradation of the chemical environment is likely to result in reduced salmonid abundance regardless of species, habitat scale or location. Thus, the pervasive nature of this control within the environment, independent of scale and location, may partly explain why it appears to exert such a significant influence over both salmon and trout, at both the fry and parr life-stage. However, it is suggested that not all impacts of hydrological connectivity and land cover are independent of location. Salmon fry abundance was negatively correlated to the presence of gravel siltation, but no such relationship was observed for trout fry (Table 6.5). It could be that this is related to differences in the scale and location of habitat occupied. Chapter Six (Section 6.3.2.1) considered relationships between gravel siltation and habitat controls within a hierarchical approach. It was suggested that, whilst siltation is predominantly explained by variations in surface hydrological connectivity risk at a catchment-scale, within areas or reaches of relatively uniform fine sediment delivery, siltation risk is related to channel slope, physical biotope and the propensity for deposition. This research has suggested that salmon typically utilise wider streams of lower gradient for spawning. This also appears to be the same environment where fine sediments are more likely to be deposited and accumulated (Soulsby *et al.*, 2001). Hence, salmon redds may be more susceptible to siltation than trout redds due to their location within the catchment.

Third, trout fry abundance showed a significant positive relationship to overhead cover which was not evident for salmon fry. It is widely acknowledged that overhead cover is an important component of juvenile trout habitat (Elliot, 1994; Heggenes *et al.*, 1999; Armstrong *et al.*, 2003), a finding that has been related to their territorial and aggressive behaviour which is considered stronger than that of juvenile salmon of the same age (Summer *et al.*, 1996; Armstrong *et al.*, 2003). Overhead cover is thought to increase visual isolation and therefore reduce aggression between fish (Valdimarsson and Metcalfe, 1998). Hence, species-specific relationships to cover appear to be related to behavioural differences in the level of territoriality expressed. However, controlled laboratory experiments have demonstrated that territorial behaviour and defence of territories with more cover may also be observed for juvenile salmon in relation to light intensity (Valdimarsson and Metcalfe, 1998) and following an increase in the perceived threat from predation (Johnsson *et al.*, 2004). It is therefore interesting that salmon fry did not show any significant relationship with the amount of overhead cover. Within the literature, it is hypothesised that this may be due to species-specific differences in feeding habits (Armstrong *et al.*, 2003) with the suggestion that increased overhead cover may reduce salmon abundance by reducing light penetration and autochthonous production (O'Grady, 1993). An alternative perspective is that these behavioural and dietary differences in relation to overhead cover may also be related in part to species-specific distinctions in the scale and location of habitat occupied and three potential explanations are suggested.

First, PCA indicated that narrow streams are typically associated with greater amounts of overhead cover than wider streams (Table 6.3). This is unsurprising as the smaller the distance between the channel banks the greater the likelihood of canopy closure above the channel. It may therefore be the case that in narrow streams, utilised by trout fry, riparian trees represent a plentiful and possibly the dominant source of cover, relative to the area of the stream bed. Hence, trout fry may have adapted to utilise overhead cover to maximise protection from predation and competition. In wider streams, riparian trees may only provide cover over a relatively small proportion of the total stream bed. Hence, salmon fry may have adapted to utilise other forms of cover that are more abundant within this scale of habitat. For example, research has found juvenile salmon abundance to be related to the amount of cover available from aquatic macrophytes (e.g. Riley *et al.*, 2006), that are typically associated with open reaches where increased light penetration supports in-stream production.

Second, and paradoxically, correlation analysis indicated a negative relationship between decreasing overhead cover and channel slope (Figure 6.5), suggesting that overhead cover is more abundant in lowland reaches. This makes sense considering the historic deforestation of the U.K. uplands together with the less amenable climate and topography for trees (Skinner and Brown, 1999). Hence, in upland locations, predominantly utilised by trout fry, a lower supply of overhead cover may cause it to become a more significant limiting factor. Relating these two points together, there may be a narrow transitional zone within the landscape, situated in the 'piedmont' zone, between the uplands and lowlands where streams are small enough to generate optimal conditions for trout in terms of channel width and overhead cover.

Third, it has been suggested within the literature that the risk of predation may depend on food availability and time spent foraging (Metcalf *et al.*, 1999 *cited in* Armstrong, 2005). As discussed above, juvenile trout may typically occupy more marginal environments, with a lower abundance of food. Trout may therefore require the greater protection provided by overhead cover as they are required to spend more time foraging. Tree cover may also provide additional food by increasing inputs of terrestrial invertebrates (Summers, 2000) and increasing nutrient availability through the provision of leaf litter (Prochazka *et al.*, 1991, *cited in* Armstrong *et al.*, 2003). Salmon fry that occupy channels with higher levels of connectivity and more abundant in-stream production may have to spend less time foraging. Their ability to hold station at higher water velocities than trout of the same size, due to larger pectoral fins (Crisp, 1996) may also increase food intake and reduce foraging time. Therefore, salmon may be less reliant upon overhead cover for food and protection. Instead, they may be more able to utilise substrate interstices to provide cover from predation due to a greater proportion of time spent sheltering. Results from the area-scale analysis suggested that in the Ullswater and Lowther Valley catchment where catchment-channel connectivity risk is very low, even in wider channels, salmon fry show a positive relationship with increasing overhead cover. It is proposed that this could be related to increased food and/or increased protection from predation and competition in an area where greater time must be spent foraging. In other words, salmon may be responding to their environment to behave more like trout. Within the literature it has been proposed that an increase in food availability is likely to make faster flowing areas of stream more available to trout because they can gain more energy to compensate for costs of holding position (Armstrong *et al.*, 2003). In such cases trout are responding to their environment to behave more like salmon. This emphasises the need to consider the impact of habitat scale, location, environmental conditions and food availability upon the behaviour of juvenile salmonids and their response to habitat. Different restoration strategies may even be required for the same species dependent upon the scale and type of habitat occupied.

Finally, correlation analysis indicated positive relationships between salmon fry and the presence of gravels and gravel siltation suggesting that salmon fry abundance/distribution is dependent upon the presence of suitable spawning substrate within the immediate vicinity. This requirement has been widely mentioned within the scientific literature and associated with the limited dispersal capabilities of fry (Armstrong *et al.*, 2003; Nislow *et al.*, 2004; Armstrong, 2005). It is interesting that at the catchment-scale salmon fry abundance appears predominantly related to variables that can be associated with the location, proximity and quality of spawning habitat (e.g. gravel siltation, channel slope, hydrological connectivity risk and gravel presence). This suggests that a major habitat bottleneck for salmon within the Eden catchment is the availability and quality of spawning habitat. This differs from the spatial pattern of trout fry abundance for which the most significant explanatory variable was overhead cover, suggesting that the major habitat bottleneck for trout may be occurring at the fry life-stage associated with the availability of cover. One explanation for the greater dependence of salmon fry on proximity to spawning habitat may be related to the scale and location of habitat utilised. It has already been suggested that the location of salmon spawning habitat within areas of lower channel slope puts salmon redds at a higher risk of siltation than trout redds. Additionally, within wider streams, spawning habitat is more likely to be spatially segregated and clustered with greater dispersal distances from one spawning site to the next. As discussed in Chapter Two (Sections 2.5.1 and 2.5.2), spawning and fry habitat are typically associated with riffles (Crisp, 1996; Summers *et al.*, 1996; Hendry *et al.*, 1997). Riffle-pool spacing is positively correlated with channel width (Leopold and Wolman, 1960), and a commonly used guide to approximate riffle/pool spacing is 5-7 times the stream width (Sear *et al.*, 2003). In wider channels, dispersal distances between riffles are likely to be greater and require swimming through deeper more extensive pool environments. This will involve greater energy expenditure and a higher risk of predation from larger fish (Einum and Nislow, 2005). Therefore, in wider channels, unless spawning habitat is available within the same riffle as juvenile habitat, it is less likely that fry will be able to migrate in to utilise the available habitat. In narrow streams, shorter dispersal distances and a reduced threat of predation could mean that trout fry are less dependent upon immediate spawning habitat proximity than salmon. Similar observations have been made for pool dwelling trout, with emigration three times higher from pools separated by short riffles (<10m in length) compared with pools separated by long riffles (>50m in length) (Lonzarich *et al.*, 2000).

7.3.2 Summary of Hypothesis (2) discussion

Untangling the complex cause and effect relationships between the types of habitat utilised and the genetic or phenotypic behavioural, anatomical and physiological responses of salmonids is outside the scope of this project. However, the findings do support the theory of niche separation (e.g. Heggenes *et al.*, 1999; Riley *et al.*, 2006) with different species evolving and developing different adaptations suited to the predominant habitat scale and type of environment in which they find themselves. Trout and salmon fry clearly occupy different scales of habitat in different locations of the Eden catchment. They also exhibit specific responses to habitat, which appear to be linked to the scale and location of habitat utilised. Salmon appear to be significantly regulated by the availability and quality of spawning habitat, together with both low and high levels of hydrological connectivity. Trout appear to be regulated at the fry life-stage by overhead cover availability and high levels of hydrological connectivity. In terms of these habitat controls, it is proposed that restoration strategies aimed at individual species will be most effective if targeted at the type and scale of habitat which they primarily utilise. For example, tree planting to increase overhead cover is likely to be most effective at improving juvenile trout populations if targeted towards narrow streams. Soil conservation to reduce siltation of redds may be more effective at improving salmon populations in wider lowland streams. In connection with this proposal, it is noted within the scientific literature that, whilst managing habitat within locations that are clearly within the habitat niche of either one species may be relatively easy, managing habitat in locations where both species are present will be more uncertain due to complex interactions between species and differences in their response to habitat (Armstrong *et al.*, 2003). For example, increasing cover may result in an increase in juvenile trout abundance leading to a reduction in salmon abundance as a result of inter-specific competition. This competitive zone is most likely to occur in the transitional landscape zone between the uplands and lowlands where habitat diversity is assumed to be highest and the balance of habitat pressures and habitat benefits optimal for both species. Other habitat controls, most notably high levels of land cover and hydrological connectivity risk, appear to influence salmonid abundance independent of species, habitat scale and life-stage. As such, their impact may be much more extensive throughout catchments requiring a wider catchment-scale approach to restoration across a larger number of habitat types and scales compared with strategies aimed at addressing species and life-stage specific habitat pressures. However, these strategies may still be targeted effectively by considering which areas of the landscape represent the greatest relative risk to water quality degradation in terms of their land cover and degree of hydrological connectivity to the channel (Lane *et al.*, 2006). It is also important to consider the type of environment within which habitat is situated in terms of the availability and dominant source of food as this is likely to influence fish behaviour and specific responses to habitat. This issue will be discussed further in the context of Hypothesis (3).

7.4 **Hypothesis (3):** *The scale of analysis will influence the relationships identified between habitat controls and salmonid abundance, or alternatively, the scale of the control will be related to the scale of its impact.*

The importance of considering the scale of investigation is widely documented within the scientific literature in relation to determining which habitat controls, at which scale, are most significant in explaining fisheries response (e.g. Allen and Johnson, 1997; Wiley *et al.*, 1997; Folt *et al.*, 1998; Armstrong *et al.*, 1998; Stauffer *et al.*, 2000; Walters *et al.*, 2003; Wang *et al.*, 2003). It has been argued that viewing the same ecological system from both a local and landscape perspective can result in conflicting conclusions (Wiley *et al.*, 1997) due to variations in the spatial heterogeneity of habitat variables and the scale at which fish respond to that heterogeneity (Fahrig, 1992 *cited in* Folt *et al.*, 1998). This has led to calls for studies to adopt a hierarchical approach, investigating phenomena at a range of spatial scales (e.g. Allan and Johnson, 1997; Armstrong *et al.*, 1998). Theories promoting the hierarchical structuring of catchments are emerging within the scientific literature (e.g. Thorp *et al.*, 2006). Yet, to date, few studies have been undertaken at more than one spatial scale (Folt *et al.*, 1998). Investigation scale is a critical issue for fisheries managers because, if fish respond to habitat differently at different scales, research undertaken at different scales will arrive at different answers in terms of the most effective approach to management. Despite this concern, reviews of the scientific literature have revealed that information gathered at one scale is frequently applied within fisheries management without considering the scale of the original sampling design (Folt *et al.*, 1998). This may explain contradictory results regarding the effectiveness of various restoration strategies.

7.4.1 Discussion of Eden catchment results in the context of Hypothesis (3)

To examine whether the scale of investigation impacts relationships between habitat and salmonid fry abundance within the Eden catchment, analysis was undertaken at both a catchment and area-scale. Unfortunately, analysis could not be undertaken at the tributary-scale as intended due to constraints in the number of electrofishing sites within individual tributaries. It should also be remembered that the sample sizes applied in the area-scale analysis are significantly smaller than at the catchment-scale which will result in greater uncertainty in the results. However, evaluation of differences in the general trends observed at each scale of investigation is still considered acceptable with the sample sizes available. The results of this comparative analysis are to be discussed here in relation to Hypothesis (3). First, were different relationships between habitat controls and salmonid performance observed at different scales of analysis? Second, was the scale of the habitat control related to the scale of its impact? The answers to these questions will then be discussed with regard to implications for effective fisheries management.

In respect of the first question, different relationships between habitat controls and salmonid abundance were identified at different scales of analysis as evident by comparing Table 6.7 (catchment-scale analysis) with Tables 6.11 and 6.12 (area-scale analysis). In particular, an increase in the significance of riparian-scale controls was observed at the area-scale compared with the catchment-scale. However, different controls were important to different extents in different areas of the catchment. Such findings have been termed 'scale inconsistencies' within the scientific literature, and their identification has been promoted as a useful tool for determining the scale over which processes controlling fisheries distribution and abundance operate (Folt *et al.*, 1998). In the Eden catchment, variations in salmonid response to habitat at different scales of investigation appeared to track differences in the spatial variability of habitat controls. Most notably, a reduction in the spatial variance of land cover and surface hydrological connectivity risk was observed as the scale of investigation contracted (Table 6.10). It is proposed that this represents a homogenisation of land cover influence, as filtered by hydrological connectivity, as the scale of investigation contracts. This causes the impacts of land cover upon in-stream conditions to become less identifiable within the spatial pattern of salmonid abundance. At the same time, the relative spatial variance in riparian and in-stream habitat controls remains high, enabling the influence of these variables to become more identifiable within the salmonid data. This effect can be hypothetically illustrated by Figure 7.4. At a large spatial scale of investigation (a) catchment-scale variables exert the greatest relative influence on salmonid abundance but at a small spatial scale of investigation (b) local scale controls exert the greatest relative influence.

In this respect, the ability to detect relationships between salmonid abundance and habitat controls is dependent upon the ability to detect spatial variability in the control (Lambert and Allen, 1999). At a small-scale, studies may obscure the overriding influence of large-scale processes on salmonid abundance simply because they do not capture enough spatial heterogeneity in the large-scale processes (Folt *et al.*, 1998). In connection with the second question proposed, these results (scale inconsistencies) indicate that catchment-scale land cover and hydrological connectivity processes operate over a greater spatial extent relative to riparian and in-stream controls, which are more localised in extent. Salmonid response was also observed to track these scales of operation.

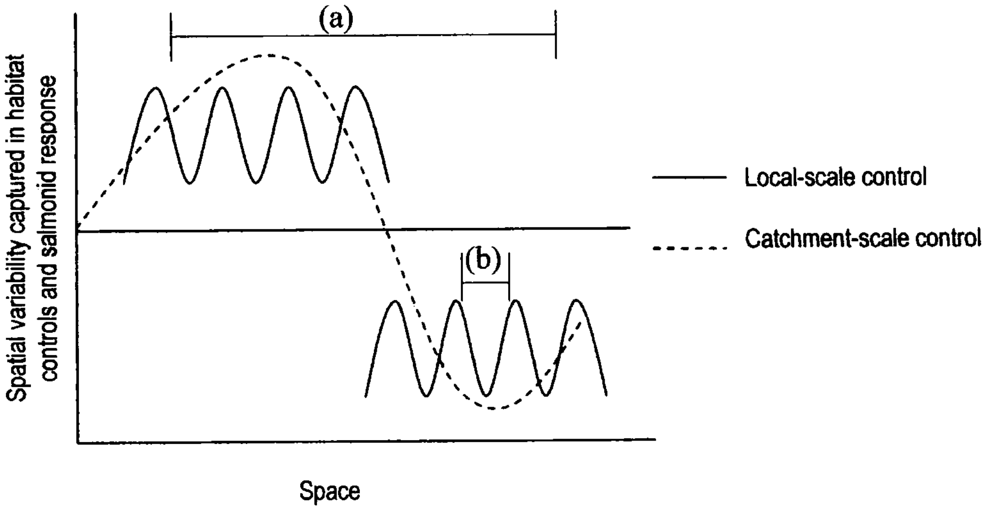


Figure 7.4: *Schematic representation of the impact of investigation scale upon the level of spatial variance captured for different scales of habitat control in relation to the response of salmonid abundance. (a) Large-scale investigation; (b) small-scale investigation.*

Similar conclusions were drawn by Lambert and Allen (1999), who studied the impact of habitat on ecological status in the River Raisin, Michigan, USA. They conducted their study in a small sub-catchment of relatively uniform land use finding in-stream habitat and local land use to be the most significant factors explaining fish response. However, they also referred to an earlier study in the same catchment which ascribed greater predictive power to catchment land use (Roth *et al.*, 1996). This study was carried out over a larger area, providing greater contrast in land use. Similar effects were also observed for relationships between habitat controls and the presence of in-stream gravel siltation. At the catchment-scale, spatial variation in siltation was most significantly explained by variation in the risk of fine sediment delivery as controlled by surface hydrological connectivity and land cover risk. However, as the scale of investigation contracted and fine sediment delivery risk became more homogenous within areas, it was channel slope and the propensity for deposition which became more significant. These results support Hypothesis (3) which proposed that the scale of the habitat control will be related to the scale at which its impact is observed both in terms of salmonid response and in-stream conditions. However, as recognised in Chapter One (Section 1.2.4) in-stream and riparian-scale controls can exert a significant influence over salmonid populations at larger scales of investigation if they are extensively distributed or exert their influence over a wider spatial extent. This is demonstrated in the results of this research which found both the percentage of overhead cover and impassable barriers to explain a significant proportion of the spatial variance in trout fry abundance at the catchment-scale.

Interestingly, a reduction in the spatial variance of catchment-scale controls within areas also corresponded with a change in the response of salmon abundance to hydrological connectivity and land cover between areas (Table 6.11). In the Ullswater and Lowther Valley area where only extreme low values of catchment-channel hydrological connectivity risk were experienced, salmon fry abundance showed a positive relationship with increasing risk. In all other areas, where only moderate to extreme high values of catchment-channel hydrological connectivity risk were experienced, salmon fry abundance showed a negative relationship to increasing risk. The precise range of hydrological connectivity risk experienced, therefore, appears to control the precise relationships that are identified and potentially the management strategies that are recommended. This is a crucial finding in relation to previous research considering the impact of landscape scale controls on fisheries performance and may help explain contradictory results over the role of land cover in structuring in-stream ecological response. It is likely that similar effects would be demonstrated for all habitat controls to which salmonids exhibit a non-linear response (Figure 7.5). Habitat variables to which salmonids exhibit a linear response will not experience this effect of changing investigation scale.

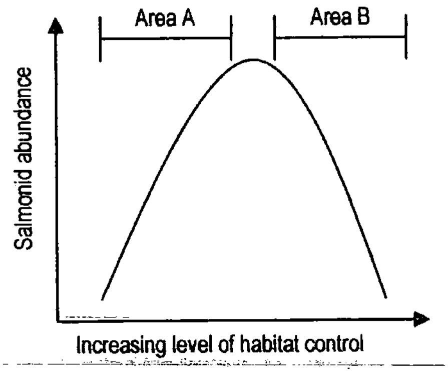


Figure 7.5: *The impact of investigation scale upon the identification of non-linear relationships between salmonid abundance and habitat controls. Area (A) would identify a positive relationship, whilst Area (B) would identify a negative relationship. The true picture would only be identified with investigation across both areas.*

Further, within areas of contrasting land cover and hydrological connectivity risk, different responses of both in-stream conditions and salmonid abundance were observed in relation to the same riparian habitat control. In the Ullwsater and Lowther Valley area (low connectivity risk) gravel siltation was related to delivery of fine sediment from bank erosion sources, and salmon fry abundance was positively related to increasing overhead cover. In the Pennine Beck area (moderate connectivity risk) gravel siltation was not related to bank erosion sources but to channel slope and salmon fry abundance was negatively related to increasing overhead cover. Analysis within these two areas appears to promote different restoration strategies, which if applied to the other area may result in either no response in the case of gravel siltation, or even a detrimental response in terms of overhead tree cover. This may help explain why contradictory responses are often observed in relation to certain habitat restoration strategies and emphasises the importance of avoiding the generalisation of management policies between river systems or even between areas within an individual river system (Armstrong *et al.*, 1998). These findings correspond with both geomorphological and ecological research presented in Chapter Two (Section 2.6). At the catchment-scale, continuum theories have been developed using landscape variables such as topography, geology, hydrological regime and climate to explain downstream trends in habitat and ecology (e.g. Schumm, 1977; Vannote *et al.*, 1980). However, research has found in-stream habitat and ecology to show little relationship to these theories at a reach scale, instead emphasising the importance of local controls leading to the development of patch dynamic theory and concepts such as physical biotopes (e.g. Padmore, 1998; Piegay *et al.*, 2000; Newson and Newson, 2000). Yet, at the small scale, difficulties have still arisen in determining the precise form of relationships between local habitat controls and salmonid abundance due to a lack of understanding of the overriding large-scale controlling processes. Within the scientific literature it is increasingly being raised, as demonstrated by this research, that the local and catchment scales are intrinsically linked and that catchments can be organised into a hierarchy of physical units where large scale processes create the template within which the small scale operates (Allan and Johnson, 1997; Armstrong *et al.*, 1998). Based on this research, it is proposed that the catchment land cover, as filtered by surface hydrological connectivity, should be included in habitat models along with variables such as geology, climate, and hydrological regime as an indicator of the broad environment in which small-scale habitat variables are operating.

7.4.2 Implications of investigation scale for fisheries management and research

The results of this research have shown that the scale of investigation does influence the relationships identified between habitat controls and salmonid abundance and therefore does affect the management recommendations that may be made. Small-scale studies may fail to capture enough spatial heterogeneity in particular habitat controls and the response of salmonids to those controls to identify significant relationships. As a result, reduced variance in either or both habitat and salmonid abundance may lead to a reduction in the explanation of variance as the scale of investigation contracts (Wiley *et al.*, 1997). This may help explain why, at an area-scale, no relationships were identified between habitat and salmon fry in the Orton/Howgill Fell area or between habitat and trout fry in the Tyne Gap area. In terms of management, this may lead to a situation where the overriding limiting factor fails to be identified, resulting in ineffective management strategies being developed (Folt *et al.*, 1998). For example, in an area of high land cover and hydrological connectivity risk such as the Caldew/Petteril area or Orton/Howgill Fell area, restoration of bankside habitat may only achieve very limited results if salmonid abundance remains constrained by poor water quality. However, without a catchment-scale approach to an investigation, this constraint may go undetected. Conversely, large-scale studies may fail to recognise the importance of small-scale, local interactions, between habitat and salmonid populations resulting in researchers missing evidence of important ecological processes. A similar effect has been observed with regard to modelling hydrological systems at a catchment-scale, where a tendency to attribute variability in calibration data to "noise" caused by sampling and analytical errors may lead to misleading results (Harris and Heathwaite, 2005). Within this research, catchment-scale PCA failed to explicitly capture variability in localised relationships between the cause of bank erosion and its severity. At the area-scale, a reduction in the spatial variability of dominant erosive processes resulted in the cause of severe bank erosion and its impact upon the environment becoming more identifiable as the level of spatial noise was reduced. At the same time, salmonid abundance exhibited a clearer response to the presence of severe erosion with a number of areas reporting a significant negative relationship between the occurrence of stock trampling, stock access and fry abundance. In terms of management, this emphasises the concept that the scale at which fish respond to habitat is intrinsically linked to the scale at which spatial heterogeneity in habitat controls and processes is expressed. To capture and understand the influence of habitat controls upon salmonid abundance, it is important to integrate investigation across spatial scales. In terms of management and research this involves major logistical issues as increasing the spatial scale of investigation whilst retaining a fine spatial resolution capable of detecting variability at the small-scale requires a dramatic increase in the number of sample sites (Folt *et al.*, 1998; Williams and Hendry, 2003). This partly explains why multi-scale studies are so rare within the scientific literature. To address this issue, this research has promoted and demonstrated the use of rapid semi-quantitative and probabilistic techniques based on new technologies for quantifying spatial variation in both salmonid populations and habitat. These techniques enable a greater number of sample sites across a wider area to be investigated. Whilst they do not provide absolute information regarding salmonid stocks and habitat quality, they are able to generate a relative picture of spatial patterns that can be interrogated to identify relationships between habitat and salmonid abundance at a range of scales. Despite the application of these new technologies, sample sizes at the area-scale were not ideal and it was still not feasible to collect the amount of data required to undertake analysis at the tributary-scale. However, as technologies develop it is hoped that these logistical issues may be addressed. As noted in Chapter One, fisheries managers may not necessarily require absolute details upon which to base decisions. Instead, it is broad-scale information indicating which factor is limiting where and at what scale that is required to prioritise and target restoration effectively.

A second issue is that studies undertaken in different locations may result in contradictory results. Stauffer *et al.* (2000) studied the effects of habitat within a catchment degraded by agriculture finding that riparian-scale variables were more significant than catchment-scale variables in determining fish assemblages. Based on these findings they concluded "that in areas of intensive agriculture, riparian protection and restoration may greatly benefit fish communities". Wang *et al.* (2003) studied the effects of habitat controls within an undegraded catchment finding reach scale variables to be more significant than catchment and riparian-scale variables in determining fish assemblages. They concluded that "local-scale habitat improvement will be most effective in watersheds that are largely undegraded and will be less effective in degraded watersheds". These two studies generate contradictory advice for fisheries managers as to the most effective approach to management. Based on a hierarchal approach and the results of this research it is likely that local-scale habitat improvement would achieve relative benefits within both areas, but that improvements would be most dramatic within the undegraded region relative to the degraded region where stocks are likely to remain constrained by agricultural degradation. Conflicting results as to the exact response of salmonids to a particular habitat control may therefore occur depending upon the larger scale within which small-scale processes are investigated. This highlights the problem of applying generalised management strategies from one location to another without consideration of the larger scale context in which they are being applied (Armstrong *et al.*, 1998).

Following a multi-scale approach to investigation, depending on which factors appear to be limiting production, it may be more appropriate to focus management actions at one particular scale, in one particular location (Armstrong *et al.*, 1998). For example, in the Eden catchment, it is proposed that improving basin-wide salmon stocks will be most effectively achieved by focusing on catchment-scale land management and hydrological connectivity to reduce gravel siltation and improve water quality. This would focus work within areas of high land cover and hydrological connectivity risk such as the Caldew/Petteril or Orton/Howgill Fell areas where management should be targeted towards those parts of the landscape which pose the greatest risk to in-stream conditions. On the other hand, riparian and in-stream habitat restoration will be most effective if targeted in those areas of the catchment that are not limited by land cover and hydrological connectivity. For example, within the Pennine Becks area, coppicing tunnelled vegetation in reaches supporting salmon fry may be beneficial. It has been suggested that advocating management of large-scale processes may be challenged in some instances because of ecosystem complexities and conflicts with other user groups (Armstrong *et al.*, 1998; McDonald *et al.*, 2004). In connection with this issue, management processes, capabilities and responsibilities may also vary across scales. For example, maintaining a crucial area of spawning habitat may be within the powers of the local landowner. Controlling diffuse pollution across catchments may be the responsibility of a government agency which has a much more restricted capacity to insist on the adoption of particular land management strategies (Armstrong *et al.*, 1998). The institutional management framework within the Eden catchment was presented in Chapter One (Section 1.3.2), illustrating the disparate nature of responsibilities and geographical remits of different organisations in relation to managing the freshwater environment. The scale of restoration possible may therefore be related to an institution's individual or collaborative remit. For example, an angling club may only have facilities to implement riparian and in-stream habitat restoration in areas where they own the fishing rights. In such cases, understanding the small-scale processes which control salmonid abundance may be more important than large-scale processes over which they have no control. It is therefore essential to understand the scale of management possible and restoration aims sought before deciding on the scale of research required to inform that management.

7.4.3 Summary of Hypothesis (3) discussion

In agreement with Hypothesis (3), the scale and location of investigation did influence the relationships identified between habitat and salmonid abundance within the Eden catchment. The scale at which fish responded to particular habitat controls was observed to track the scale at which spatial variation in the control was most clearly expressed. Additionally, the response of in-stream conditions and salmonid abundance to habitat was observed to depend upon: (1) the state of large-scale processes within which small-scale processes were operating; and (2) the range of habitat variability over which processes were studied in the case of non-linear responses. These findings were then discussed in terms of implications for fisheries management and research emphasising the need to adopt a hierarchical approach to investigation which considers the scale of restoration that is desired and achievable within individual catchments.

7.5 Effective approaches to habitat restoration: The Eden catchment case study

A major advantage of the approach taken throughout this research is that spatially distributed habitat data were collected for the majority of habitat variables. This enables the findings identified through multivariate analysis and discussion of the three hypotheses to be extrapolated and effectively converted to management strategies targeted at those locations where they are likely to be most beneficial. An additional advantage is that the habitat data can be readily visualised in map and photographic form using GIS, creating a powerful tool for justifying restoration strategies to the public and engaging local communities, landowners and funders in restoration efforts. To demonstrate the potential for achieving this, the example of developing a targeted catchment-scale restoration strategy for trout within the Eden catchment is presented.

Catchment-scale land cover

The impact of increasing land cover risk, as filtered by hydrological connectivity, was identified as a significant factor limiting both trout fry and trout parr at the catchment-scale. The pervasive nature of this control within the environment appears to limit salmonid populations regardless of life-stage, species, habitat scale or location. Addressing this issue should therefore be a major priority of any fisheries management plan within the Eden catchment, across all habitat types. It has been raised that addressing large-scale processes may be problematic due to the amount of change required and conflict with other user groups (McDonald *et al.*, 2004). One of the benefits of considering land cover impacts in the context of hydrological connectivity, as modelled using the *SCIMAP* approach, is that large-scale changes to land management may not be required

(Lane *et al.*, 2006). By focusing on delivery pathways, it may be possible to use strategies such as buffer zones to reduce hydrological connectivity or small-scale land cover change at discrete highly sensitive locations increasing the environment's resilience to diffuse pollution (Burt, 2001). In this respect, diffuse pollution may be considered a misnomer, as it actually involves a large number of point sources (well connected risks) distributed throughout the environment (Lane *et al.*, in review). A number of approaches to restoration could be adopted. Reaches with potentially greater abundance that are currently impaired by land cover could be prioritised in the first instance. Alternatively, reaches of existing high abundance with a high risk of future degradation due to high levels of hydrological connectivity could be protected to help retain their high productivity (Pess *et al.*, 2002). Here it is assumed that the first approach is to be adopted. To prioritise those locations of the catchment where land cover as filtered by hydrological connectivity has a high likelihood of limiting trout stocks, a map of the *SCIMAP* delivery index weighted by land cover (Section 5.2.1) has been classified in accordance with the response of trout fry abundance at the catchment-scale (Figure 7.6). Classification was based on graphical inspection of the relationship between the delivery index weighted by land cover and trout abundance, an arbitrary delivery index threshold was selected above which trout fry populations appear to be significantly constrained (Figure 7.7). A number of priority areas for restoration at this scale are immediately identifiable.

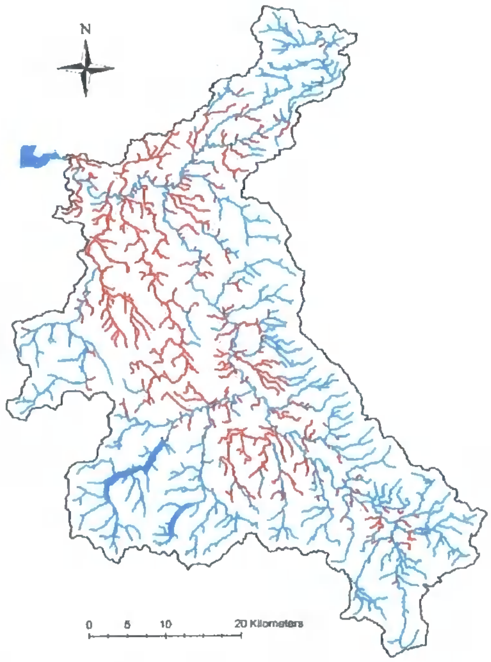


Figure 7.6: *Priority areas for restoration (shown in red) aimed at addressing the impact of land cover as filtered by hydrological connectivity upon trout populations. The SCIMAP delivery index weighted by land cover has been classified in accordance with the response of trout fry abundance to hydrological connectivity.*

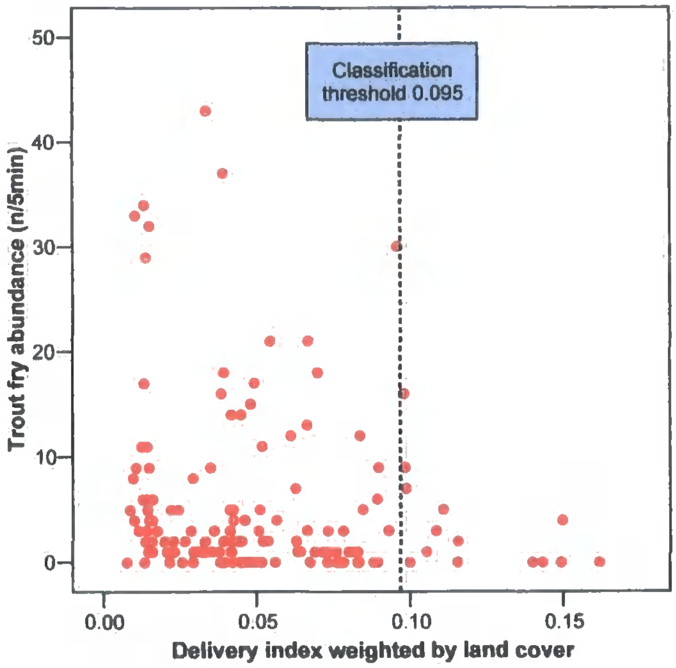


Figure 7.7: *Selection of a classification threshold for the delivery index weighted by land cover*

Riparian and in-stream habitat restoration

Comparison of relationships between habitat and trout abundance at different life-stages suggested that the major habitat bottleneck, in terms of density-dependent regulation, is occurring at the fry life-stage within the Eden catchment in relation to the amount of overhead cover available. Spatial variation in trout parr abundance showed a greater association with trout fry productivity than with habitat, a finding that was related to greater dispersal capabilities and adaptability at this life-stage. The significance of overhead cover was restricted to trout fry and related to the species-specific scale and location of habitat which they occupy. Based upon these observations, it is proposed that riparian habitat restoration should be prioritised towards trout fry habitat (riffles within narrow streams), concentrating on strategies aimed at increasing overhead cover (e.g. tree planting and stock exclusion fencing). By targeting this specific habitat type, resources for habitat restoration can be utilised more efficiently and economically to deliver maximum benefits across a wider area. Figure 7.8 highlights those areas of the Eden catchment where overhead cover is less than 25%. Due to the uncertainties of managing habitat within reaches where salmon and trout niches overlap (Armstrong *et al.*, 2003), only those reaches that are known to support trout alone or from habitat data are assumed to only support trout have been highlighted. It is important to recognise that the data presented are limited with respect to the area covered by the aerial photographs. As such, further walkover surveys of small tributaries may be required to evaluate the exact location and extent over which this strategy should be applied. Within selected areas, a number of approaches to prioritise restoration could be adopted. Reaches with extensive lack of cover (e.g. greater than 500m) could be targeted. Alternatively, only reaches where land cover is not limiting could be targeted as it is here that habitat restoration is likely to have the greatest impact (Wang *et al.*, 2003). Site visits within identified reaches should then be undertaken to identify whether trout fry habitat is present and to what extent, further concentrating restoration efforts effectively. Impassable barriers were also highlighted as having a significant control over trout fry populations at the catchment-scale and stocks may benefit from either their removal or the easement of fish passage around them. Again, it may be most effective to target this work within areas where land cover is not limiting and where a large area of upstream habitat is impacted.

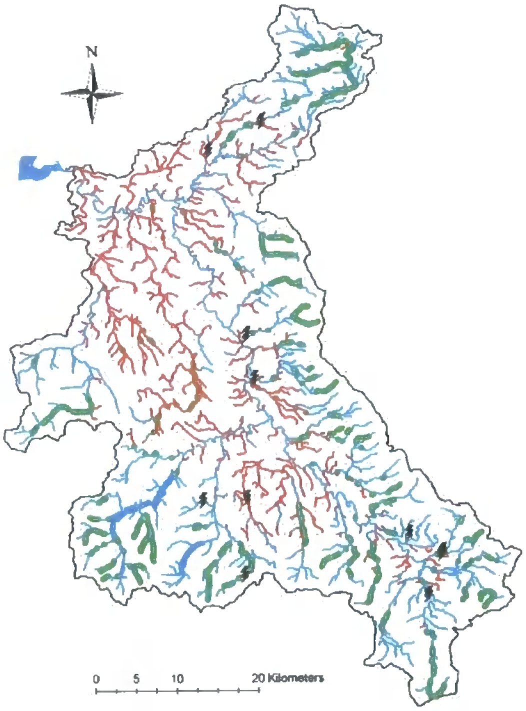


Figure 7.8: *Priority areas for restoration strategies aimed at increasing the amount of overhead cover (shown in green) and barrier removal (lightning bolt) within juvenile trout supporting streams. The areas identified are restricted by the extent of aerial photograph coverage. This has been combined with Figure 7.5 highlighting priority areas for addressing catchment-scale land management.*

Area-scale restoration

Different relationships between habitat and trout fry abundance were observed dependent upon the scale of investigation. As investigation scale contracted, the influence of catchment-scale land cover and percentage overhead tree cover declined. Instead, the presence of bank erosion due to stock trampling and the presence of gravel substrate were highlighted as important in explaining spatial variation in trout fry abundance. It has been discussed that these differences are related to the spatial scale at which heterogeneity in the habitat control is most clearly expressed. Based on these results, it is proposed that restoration strategies aimed at reducing stock access be considered at a more local-scale, in areas where catchment-scale land cover and overhead tree cover are not limiting trout fry populations. One such area within the Eden catchment where these strategies may be effectively applied is the Pennine Becks area. Here area-scale analysis suggested that the presence of erosion due to stock may be the most significant control over trout fry populations. Again, reconnaissance surveys should be used to identify the location and extent of trout fry habitat.

A final point to remember when developing effective restoration strategies is that organisms require an access corridor through which to recolonise restored habitat. This reflects the ecological perspective of connectivity which controls the ease with which organisms can move between habitat patches (Moilanen and Hanski, 2001). It may be futile, unless artificially stocked (e.g. Aprahamian *et al.*, 2003), to restore habitat within a reach that remains surrounded by degraded habitat with no accessibility for current salmonid populations. The most effective strategy may therefore be to restore degraded habitat where small populations are in existence or in corridors extending from locations of good habitat and high abundance to areas of poor habitat and low abundance.

7.6 Chapter summary

A key output of this research is a spatially-structured hierarchical GIS that allows hypotheses regarding habitat controls and salmonid performance to be tested using multivariate statistics. The aim of this chapter was to discuss the results of this analysis, first in relation to the three hypotheses formulated in Chapter Two and second, in relation to developing effective approaches to habitat restoration using the Eden catchment as a case study. Results have been presented showing that management decisions can only be undertaken with a full account of spatial scale in relation to life-stage and species-specific habitat requirements, as different habitat controls are relevant in different locations and over different scales. Whilst no single scale of habitat control is solely responsible for explaining salmonid abundance and distribution within the Eden catchment, the degree to which species are related to a particular scale of process may be in part related to the scale of their distribution and mobility, the scale of habitat which they occupy and that scale of analysis.

Specifically, testing of Hypothesis (1) suggested that relationships between habitat and salmonid abundance are structured according to the potential for dispersal at different life-stages. At the fry life-stage, limited dispersal capabilities appear to result in population regulation by habitat induced effects. Conversely, spatial variability in the abundance of more mobile salmonid parr appears more significantly explained by recruitment from fry than by habitat controls. Based on this, it is suggested that habitat restoration will be most effective if focused at fry and spawning habitat (e.g. riffles) within the Eden catchment as this is where the greatest habitat bottleneck is occurring. With regard to Hypothesis (2), significant species-specific differences in response to habitat controls were observed, particularly at the fry life-stage. This has been related to distinct differences in the scale and location of habitat utilised by each species resulting in the proposal that specific restoration strategies will be most effective if targeted towards specific habitat scales in specific locations of the catchment. The exception to these two findings is the impact of increasing linear land cover risk as filtered by hydrological connectivity upon salmonid abundance. This control was found to be significantly related to the spatial pattern of salmonid abundance regardless of life-stage, habitat scale and species. As such, a wider catchment-scale approach to restoration across a wider range of habitat scales and types may be required to address the issue of catchment-scale land cover in comparison with more life-stage and species-specific habitat controls. However, the use of hydrological connectivity may be an effective mechanism for targeting such restoration without the need for large-scale land cover change that would be impracticable to achieve. Finally, in respect of Hypothesis (3), the scale of investigation did impact upon the relationships identified between salmonid abundance and habitat. The scale at which fish responded to particular habitat controls appeared to be related to the scale at which spatial variation in the habitat control was most clearly expressed. Further, at the area-scale, the exact response observed to certain habitat controls appeared to be related to the range of land cover and hydrological connectivity risk experienced. In this respect, large-scale processes were controlling the environment within which the small scale operated (Armstrong *et al.*, 1998). This highlighted the dangers of applying aspatial results uniformly within habitat management strategies across an entire catchment, and emphasised the need for a spatially explicit and hierarchical approach to research considering the scale of restoration desired. In conclusion, it is considered that recognition of scaling issues is essential to the development of effective approaches to river management.

Chapter Eight: Conclusions

The aim of this chapter is to revisit the aims and objectives of the thesis and to summarise results in relation to these aims, discussing their implications in terms of the wider context of fisheries management and highlighting areas for future research.

8.1 Review and critique of thesis aims and objectives

The aim of this research was to couple recent advances in remote sensing, Geographical Information Systems, environmental modelling and ecological surveying techniques with current ecological understanding of habitat controls on salmonid populations, to develop a more effective approach to prioritising habitat restoration.

Chapter One (Section 1.2.1) recognised that, whilst habitat is perceived to exert a significant influence over salmonid population dynamics, habitat restoration strategies have frequently failed to achieve the widespread improvements desired. One hypothesis for this is that they fail to capture fully the influence of scale over relationships between habitat controls and salmonids in space and time. A need was identified to develop a more effective approach to informing and prioritising habitat restoration which incorporates the issue of scale. A number of scaling issues considered to be relevant to the management of salmonid fisheries were identified and discussed in Chapter One (Section 1.2) including: (1) the changing habitat requirements of salmonids throughout their life-cycle; (2) the mobility of salmonids and potential for dispersal at different stages of their life-cycle; (3) the range of spatial scales at which habitat controls may operate (e.g. catchment, riparian and in-stream); and (4) the importance of investigation scale (e.g. catchment, area, tributary) in controlling the relationships that are identified. Whilst these scaling issues are widely recognised within the scientific literature (e.g. Wiley et al., 1997; Stauffer et al., 2000; Wang et al., 2003), and hierarchical theories and models have been proposed to conceptualise them (e.g. Allan and Johnson, 1997; Armstrong et al., 1998; Thorp et al., 2006), there have been few actual studies that investigate the impact of scale within the context of real environmental systems (e.g. Folt et al., 1998). The prime constraint has been the ability to gather the data required at a resolution relevant to each scale of control and analysis. The challenge for this thesis was to develop an approach capable of investigating the importance of scale whilst remaining practical enough for fisheries managers to apply. Thus, the research was divided into three distinct sections based on three main research objectives:

Objective 1: *To review and to synthesise current understanding of in-stream, riparian and catchment-scale controls on freshwater habitat throughout the salmon and trout life cycle.*

Objective 2: *To employ recent advances in remote sensing, GIS, and environmental modelling, to identity, to develop and to validate tools for quantifying salmon and trout habitat at the catchment-scale, appropriate to each habitat control and scale of control.*

Objective 3: *To use the data acquired under Objective (2), to investigate hypotheses identified through Objective (1) regarding relationships between habitat controls and salmonid populations, and to discuss the results in the context of effective approaches to habitat restoration.*

The research aimed to be generic but was developed and applied to the River Eden catchment, Cumbria, UK, with particular focus on, Atlantic salmon and brown trout.

8.1.1 Objective 1: *To review and to synthesise current understanding of in-stream, riparian and catchment-scale controls on freshwater habitat throughout the salmon and trout life cycle.*

Critical to the success of this research was the ability to capture the ranges of scales at which controls on salmonid habitat can occur (i.e. in-stream, riparian and catchment scale) and to identify from the outset a set of habitat controls from across each of these scales that are perceived to be ecologically relevant in terms of salmonid performance. However, it was recognised that current understanding regarding relationships between habitat controls, in-stream conditions and salmonid populations across these scales is divided amongst a number of distinct, but all too often disparate, scientific disciplines. Ecological research has focused on population dynamics and the relationship between the environment and population regulation. Fisheries science has concentrated on understanding the in-stream habitat requirements of salmonids at different stages of their life-cycle. Hydrological and geomorphological research has studied the factors and processes operating within the riparian zone and wider catchment that influence in-stream physical, chemical and biological conditions. There was therefore a need to synthesise current understanding from across these three disciplines into a single hierarchical framework which could be used to guide further evaluation, and to aid the formulation of integrated hypotheses regarding the impact of scale upon relationships between habitat controls and salmonid populations. This was the basis of Objective (2) which was addressed in Chapter Two of the thesis.

The DPSIR (Driver – Pressure – State – Impact – Response) framework was identified as a potential mechanism for integrating and synthesising a large volume of information and research from across disciplines. Typically used for integrating science with socio-economic policies, the approach was adapted here to conceptualise relationships between controls at the large-scale (Drivers), within which controls at a smaller-scale exert their influence (Pressures) upon in-stream conditions and salmonid habitat at different stages of the life-cycle (State), ultimately affecting salmonid abundance through a variety of mechanisms (Impacts). Restoration strategies (Responses) can then be linked back to earlier stages in the framework to consider their sustainability and potential impact upon salmonid populations. The DPSIR framework proved to be highly successful at integrating research from across disciplines by identifying commonalities. For example, under the 'State' category, biological habitat requirements understood by ecologists were readily linked to abiotic components of the environment, the controls of which hydrologists and geomorphologists understand. The main advantage of this approach was its hierarchical structure which allowed the role of habitat controls and processes operating at different scales to be explicitly captured within a single, simple, but holistic framework. This review and synthesis enabled three hypotheses to be formulated regarding the impact of scale upon relationships between habitat and salmonid abundance/distribution at different levels within the DPSIR framework:

Hypothesis (1): Relationships between habitat and salmonid abundance/distribution are structured by life-stage according to the level (scale) of mobility and potential for dispersal at each life-stage.

Hypothesis (2): Relationships between habitat and salmonid abundance are species and location specific relating to the scale of habitat occupied by different species.

Hypothesis (3): The scale of analysis (e.g. catchment, sub-catchment, reach) will influence the relationships identified between habitat controls and salmonid abundance/distribution, or alternatively the scale of the control will be related to the scale of its impact.

In addition, the review and synthesis of current understanding helped identify a selection of ecologically relevant habitat controls for further investigation at the catchment (Driver), riparian (Pressure) and in-stream (State) scales.

8.1.2 **Objective 2:** *To employ recent advances in remote sensing, GIS, and environmental modelling, to identity, to develop and to validate tools for quantifying salmon and trout habitat at the catchment-scale, appropriate to each habitat control and scale of control.*

To address the hypotheses formulated under Objective (1), a hierarchical approach to analysis was required which extended from in-stream processes at the individual tributary scale to landscape processes at the catchment scale, across both trout and salmon and both fry and parr habitat. As discussed throughout the thesis, one of the main obstacles to this type of research is data availability. Traditional surveying techniques for both habitat and salmonid population data have focused on gathering detailed information at small spatial scales, proving prohibitively costly and time consuming at the larger spatial scales required here (Wiley et al., 1997). In addition, conducting multi-scale studies involves major logistical issues as increasing the spatial scale of investigation whilst concurrently retaining a fine enough spatial resolution to detect variability at the small-scale requires a dramatic increase in the number of sample sites (Folt et al., 1998; Williams and Hendry, 2003). The purpose of this objective was to evaluate the contribution that emerging technologies and approaches can make to the collection of this data and to assess whether assessment of salmonid habitat across entire catchments is feasible. Thus, Chapter Three presented the broad-scale data requirements and datasets available. Chapter Four considered the use of high resolution digital aerial photography and topographic data for providing information on in-stream and riparian-scale habitat controls. Chapter Five focused on the potential of hydrological connectivity, specifically the *SCIMAP* model, as a framework for investigating the impact of catchment land cover on salmonid abundance, in-stream habitat and water quality. The aim was not to evaluate whether these tools could provide completely accurate descriptions of habitat condition but rather to consider their ability to identify relative pressures on salmonid habitat from one location to the next through space. It is this ability to quantify relative risk which is most important to fisheries managers in enabling them to prioritise one location over another in respect of habitat restoration.

In terms of the success and limitations of the three methodologies applied in Chapter Four, Table 4.18 provided a summary of which habitat variable, traditionally collected by a walkover survey, can or cannot be measured using high-resolution digital aerial photography and topographic data. Visual assessment of 20cm resolution aerial photography proved extremely successful, especially for mapping riparian habitat controls such as overhead cover and bank erosion due to stock access (Tables 4.3 - 4.6). Both these factors were found to contribute significantly to the explanation of spatial variation in salmonid abundance at both the catchment and/or area-scale.

As such, the technique and data collected should be highly beneficial to managers looking to quantify riparian habitat restoration needs within the Eden and other agricultural catchments. However, factors such as channel substrate and channel modification were less identifiable, due to their smaller scale (substrate) or requirement for an oblique not vertical view (modification). In a more urbanised catchment, where these factors are more likely to be limiting to salmonids as a result of greater channel modification, the technique may be of less benefit. A further limitation was that no photography was captured for streams smaller than 2m wide due to resolution limitations. Throughout this thesis, and in combination with other research by the Eden Rivers Trust, it has become apparent that streams of this size may be particularly important for supporting juvenile trout. Whilst electrofishing surveys targeted streams of this size in 2006 (Brown, 2006b) there were no accompanying images for multivariate analysis. Thus, the analysis of relationships between habitat and trout fry may have therefore been slightly restricted. The technique was also compared to a traditional walkover survey in terms of practicality and cost (Section 4.2.5). In this respect it was considered an excellent alternative to ground surveying in large catchments. The advantage of providing a continuous and permanent picture allows managers to revisit the images following multivariate analysis to determine the precise extent and location of pressures identified as significant. Where restrictions were encountered in reaches of dense vegetation or very small streams a ground survey may still be the preferred option. However, aerial surveying provides baseline information that enables walkover team resources to be prioritised more effectively.

Automated image processing techniques were also applied to the aerial photography (Section 4.3) to extract relative depth information to aid the classification of in-stream habitat type (e.g. fry, parr habitat). Results for individual photographs were extremely promising. However, illumination variations between images severely restricted the technique's applicability to the classification of image mosaics at the catchment-scale. As such, it was not considered a viable alternative at the current time to the traditional walkover survey for mapping in-stream habitat at the catchment-scale. This outcome did not adversely impact the progression of this thesis as all fry electrofishing surveys were undertaken within suitable riffle habitat and for all parr surveys the percentage of riffle and pool was recorded. It is more of an issue for managers looking to target remotely particular restoration strategies at particular life-stage specific habitat types following the outcome of multivariate analysis. However, as technologies advance and higher resolution data become more readily available, it is likely that current limitations associated with both this and the virtual walkover methodology will be overcome resulting in an exciting prospect for the future of salmonid habitat assessment. Even during the life of this project, huge advances have taken place with 3-5cm resolution true colour digital aerial photography becoming commercially available. Future research should continue to evaluate the contribution such data can make to the application of salmonid habitat assessment.

Channel slope was identified within the literature as a useful 'surrogate' variable for characterising in-stream conditions (e.g. Sear et al., 2003; Walters et al., 2003), major advantages being its objective measurability, its continuous nature, and its relative stationarity through changing flow conditions (Section 4.4.5). To assess the potential for gathering catchment-scale information on channel slope remotely, a methodology was tested for extracting channel slope and further, physical biotope, from the 5m NEXTMap Great Britain ™ DTM (Section 4.4). Results indicated that a good representation of channel slope could be generated (Table 4.12 & 4.13) and that the classification of biotope was only marginally impaired by using remotely sensed channel slope compared with that measured in the field (Tables 4.15 & 4.17). However, uncertainty was high and overall the technique was considered to be more suited to broad-scale, catchment-wide assessment of habitat availability than the detailed analysis of habitat pattern within specific reaches. Multivariate analysis revealed that both channel slope and physical biotope were significantly, but oppositely, related to the abundance and distribution of salmon and trout fry. Therefore, the ability to determine these variables remotely, even if only broadly, should be a useful tool for fisheries managers looking to target species-specific restoration strategies towards appropriate locations of the catchment. Again, as technology advances, further assessment of this tool's potential should be undertaken.

One of the most successful aspects of this research has been the application of the *SCIMAP* model and the concept of hydrological connectivity as a framework for evaluating the influence of catchment land cover on salmonid populations. Despite a high level of spatial noise within the salmonid data, a striking structuring of salmonid populations was observed in relation to the level of land cover risk as filtered by topographically controlled hydrological connectivity. This relationship was found to be significant at both the fry and parr life-stage and for both salmon and trout, although different specific responses were exhibited at extreme low levels of risk. The migratory behaviour of Atlantic salmon also appeared related to this risk. Results suggested that land cover can exert an extensive impact on the aquatic environment that is independent of habitat scale, type and location, emphasising the importance of considering landscape-scale controls in studies of salmonid populations. The degree of land cover and hydrological connectivity risk was also observed to influence relationships between salmonids and habitat controls, such as the impact of overhead cover at smaller spatial scales. Lack of consideration of such factors may be one of the reasons why habitat models such as HABSCORE, which assume pristine water quality, have to date, not realised wide ranging success. One of the main advantages of the *SCIMAP* approach is that by considering connectivity it adds functional significance to the relationships between land cover, in-stream habitat and ecology, explaining why observations of land cover significance may vary between locations (Meador and Goldstein, 2003). Connectivity risk can either have a positive or negative impact upon salmonids dependent upon what the channel is connecting to. As such, it is neither land cover nor hydrological connectivity alone which determines impact but their combination. In accordance with filter/resistance theory (Brunsden, 1993, *cited in* Burt, 2001), this approach allows managers to identify which parcels of land are most sensitive or resilient to transmitting land cover impacts to the channel as a result of their hydrological connectivity. This enables managers to prioritise restoration activities towards discrete, highly sensitive, areas of the landscape avoiding the need for large-scale land management change (Lane et al., 2006). Further, by focusing on delivery pathways it may be possible to use strategies such as buffer zones to reduce hydrological connectivity and increase filter resistance in areas where it is low (Burt, 2001). The main drawback to this approach is that it does not specifically identify the mechanism by which land cover influences salmonid abundance. Within the thesis, several potential mechanisms were inferred including the delivery of fine sediment to redds and the impact of nutrients on water quality and food availability, but, it was impossible to discriminate between mechanisms. However, it is recommended that the *SCIMAP* model could be used as a tool to identify study sites for further research into the impact of fine sediment, nutrient and agricultural chemical delivery on salmonids. To date, the majority of such studies have been undertaken in the laboratory and have failed to replicate results in the field (e.g. Lower and Moore, 2003; Waring and Moore, 2004). Hydrological connectivity may provide a framework by which to expand this research in the field case.

Overall, the assessment of salmonid habitat controls across entire catchments is considered to be becoming more feasible through the application of technological advances as described above.

8.1.3 **Objective 3:** *To use the data acquired under Objective (2), to investigate hypotheses identified through Objective (1) regarding relationships between habitat controls and salmonid populations, discussing the results in the context of approaches to habitat restoration.*

To achieve Objective (3) an approach was required that was capable of collating and interrogating a large quantity of data, whilst at the same time preserving the explicit representation of spatial scale. This was the focus of Chapter Six. In the first instance, GIS processing proved exceptionally successful in enabling data to be co-registered and organised within a single hierarchical database. The creation of this database was a particularly powerful tool, allowing relationships between habitat and salmonid populations to be analysed at a variety of scales. Multiple regression and binary logistic regression were successfully applied to identify those habitat controls that explained a significant proportion of the spatial variation observed in salmonid abundance and distribution at different scales. An initial obstacle to this analysis was the spatial autocorrelation between many of the individual habitat controls. To overcome this issue, Principal Components Analysis (PCA) was applied to reduce the dataset to a suite of independent habitat factors. The use of PCA, together with correlation analysis, also facilitated investigation of relationships between habitat variables at different spatial scales. This proved especially fruitful, yielding valuable information regarding changing relationships between habitat variables as the level of spatial variability expressed and captured changed at different spatial scales of investigation. Relationships between habitat and salmonid abundance were subsequently observed also to track these changes. Understanding the influence of spatial scale on the ability to capture spatial variability in the habitat controls significantly aided the interpretation of salmonid response to habitat at different scales. Therefore, it is recommended that future studies also consider the hierarchical structuring of relationships between habitat and salmonid performance in the context of hierarchical structuring between the habitat variables themselves. The two-species approach, analysing both salmon and trout data, was also particularly useful. Identifying differences between the two species helped to identify and evaluate relationships observed for each species. For example, considering different species responses to hydrological connectivity in the context of differences in feeding habits helped to evaluate why salmon may show a non-linear response to hydrological connectivity.

The main limitation associated with this objective was that the spatial resolution of salmonid data available was insufficient to facilitate analysis at a tributary-scale and may have also impinged on the certainty of relationships identified at the area-scale. To address this issue, this research promoted and demonstrated the use of rapid semi-quantitative and probabilistic techniques based on new technologies for quantifying spatial variation in both salmonid populations and habitat. Whilst these enabled a considerable number of sample sites across a wide area to be investigated, the resolution was not fine enough to capture variability at the smallest scale, again highlighting the logistical difficulties of undertaking multi-scale projects. However, as discussed in Chapter Seven, the scale of research should be selected according to the scale of management it is primarily aiming to inform. This research was designed to inform management decisions and restoration of salmonid habitat at the catchment-scale, as required by Eden Rivers Trust, and data collection was designed as such. Following the results of the multivariate analysis, a number of sub-catchments have been prioritised for restoration projects by the Trust. In summer 2007, electrofishing is to be targeted within these tributaries. This will offer the opportunity to investigate relationships between habitat and salmonids within these discrete sub-catchments to inform management decisions further. In this way, catchment-wide studies can be used as a baseline by which to target detailed investigation effectively at smaller spatial scales. It is also important to remember, as highlighted in Chapter Seven, that whilst empirical analysis may identify significant relationships between variables, this only represents association not causation. However, it is a useful approach for identifying relationships and forming hypotheses that can be tested through subsequent research.

8.1.4 Summary of thesis objective review

Through the synthesis of current understanding regarding habitat controls on salmonids and the application of the DPSIR framework, Objective (1) provided a conceptual framework and hypotheses for the thesis. Objective (2) then applied technological advances in remote sensing, GIS, environmental modelling and ecological surveying to provide the data required to analyse relationships between habitat controls and salmonids within this framework. Finally, Objective (3) identified and demonstrated an effective approach to collating and analysing data whilst recognising the issue of scale. The ability of these three objectives to deliver the thesis aim is now reviewed by summarising the research findings in terms of their ability to inform and to develop effective approaches to prioritising habitat restoration.

8.2 Delivery of thesis aim – the development of an effective approach to prioritising salmonid habitat restoration.

The results of the thesis were discussed in Chapter Seven. This led to the conclusion that effective prioritisation of habitat restoration can only be achieved by taking a full account of spatial scale in relation to life-stage and species habitat requirements, as different habitat controls are relevant in different locations and over different scales. For restoration to be effective, it must be targeted at those habitats and locations where it will achieve maximum benefits. In addition, by reducing the number of specific habitats to which certain strategies are applied, limited resources can be maximised to deliver restoration across a wider area. In terms of targeting restoration towards specific habitats, testing of Hypothesis (1) showed that relationships between habitat and salmonid abundance are in many cases life-stage specific, structured according to the potential for dispersal at each life-stage. In terms of the DPSIR framework, the 'impact' of habitat on salmonids may be mediated by their ability to disperse away from localised pressures and avoid regulation by mechanisms such as competition. However, not all 'impacts' can be mediated by dispersal. For example, regulation as a result of toxicity and impacts of water quality on fish health may occur regardless of dispersal capabilities. This was supported by the observation that land cover and hydrological connectivity risk were significantly related to salmonid abundance at both the parr and fry life-stage. Within the Eden catchment, habitat was found to exert a greater influence at the less mobile fry life-stage with parr more dependent upon recruitment. It was therefore recommended that the most effective approach to habitat restoration is to target fry (e.g. riffle) habitat. Further, different responses to the same control ('pressure') were observed at different life-stages (e.g. the relationship between salmon and overhead cover). This reinforced the danger of applying aspatial results across a whole catchment, as certain strategies (e.g. coppicing) may not only be ineffective but they may actually be detrimental if targeted at the wrong habitat type. Similarly, testing of Hypothesis (2) indicated that relationships between habitat and salmonid abundance may be species specific, particularly at the fry life-stage when trout and salmon occupy distinctly different scales of habitat within different locations of the catchment. The particular 'pressures' experienced by fish may, in many instances, be related to the scale and location of habitat utilised and their particular behavioural and physiological adaptations to that habitat scale. Consequently, certain restoration strategies may be most effective if targeted at particular species in particular locations of the catchment. Again this highlights the dangers of applying aspatial results catchment-wide without considering the scale of habitat and species to which they are applied. Alternatively, the impact of land cover and hydrological connectivity was again observed to impact salmonid abundance irrespective of species and the scale of habitat occupied. This potentially explains why it has such an overriding influence on populations and raising the issue that a broader-scale restoration approach may be required to deliver effective restoration in respect of this pressure. Finally, the testing of Hypothesis (3) showed that the scale of research impacts its ability to identify limiting factors dependent upon the scale at which spatial variability is observed for each control and the scale at which fish can respond to that variability. The 'pressure' exerted by certain habitat controls and environmental requirements of salmonids 'state' were observed to change in response to changes in catchment-scale land cover ('Drivers'). It was therefore suggested that identifying limiting habitat controls and understanding their spatial influence can only be achieved effectively through a hierarchical multi-scale approach which considers the scale of restoration desired.

In conclusion, salmonids and scales are intrinsically linked and effective prioritisation of habitat restoration can only be achieved with consideration of scale. This research has identified: (1) the DPSIR model as an effective framework within which to synthesise research from across disciplines and consider relationships between habitat and salmonids at different scales; (2) a range of effective tools for gathering information on salmonids and habitat at a range of spatial scales, illustrating that it is feasible to assess salmonid habitat across entire catchments; and (3) an effective approach to analysing that data and identifying relationships from which restoration strategies may be effectively prioritised.

8.3 Wider context and recommendations for future research

This research has specifically focused on the impact of in-stream, riparian and catchment-scale controls upon salmonid habitat and salmonid abundance/distribution. However, as discussed in Chapter Two (Section 2.3), habitat is not the only hypothesis for salmonid population decline. It is therefore important to recognise the role of habitat in the context of these different hypotheses which are ultimately all intertwined to form the salmonid ecosystem. However, as noted in Section 2.3.7, habitat is likely to play an important role in determining the influence of other controlling factors on salmonids and should be taken into account when determining their impact. In other words, just as catchment land cover and hydrological connectivity were observed to be a large-scale process within which small-scale habitat processes operate, habitat may form the template within which other controlling factors such as predation, exploitation, disease and climate are expressed. This research has been concerned with the impact of spatial scale on relationships between habitat and salmonids. It is also important to recognise the role of temporal scale and variations through time as a result of natural fluctuations in populations, the impact of extreme events and more long-term change in response to changing habitat and climatic conditions. In addition, this research focused solely on the River Eden catchment. Just as different scales of analysis within the Eden catchment identified different relationships between habitat and salmonid abundance, it is important to remember that different relationships may also be observed in different catchments. It is therefore recommended that restoration strategies be drawn up on a catchment-specific basis and that care is taken when applying management recommendations based on research undertaken at different scales or in different locations.

In terms of the future, a number of potential avenues for further research have been identified and are outlined below:

- The impact of restoration strategies on salmonid populations ('response') should be monitored within a hierarchical framework to evaluate their success and to inform effective management strategies further. It is important to remember that the natural recovery of ecological systems will take time and appropriate monitoring should be continued over the medium and long-term. In particular, systems should be monitored at regular intervals to assess whether habitat restoration has resulted in new habitat bottlenecks occurring at different life-stages.

- It would be interesting to evaluate the role of hydrological connectivity further in explaining the spatial variation of salmonids. In this regard, applying the *SCIMAP* framework within a pristine environment in association with assessment of food resource availability, growth rates and the behavioural responses of salmonids throughout their life-cycle could be valuable. There may be natural spatial variability in salmonid populations due to hydrological connectivity but this research cannot separate this from land use effects.

- The *SCIMAP* model and hydrological connectivity may also provide a framework by which to extend laboratory-based research into the impact of various chemicals on salmonids to a larger-scale within natural environments, by helping identify potential study sites where the delivery of chemicals is likely to occur. This research would also help to identify and clarify the exact mechanisms by which land cover exerts an influence over salmonid populations.

- Also with regard to hydrological connectivity, it is suggested that catchment land cover as filtered by surface hydrological connectivity be included in habitat models along with variables such as geology, climate, and hydrological regime as a predictor of the environment in which small scale habitat variables are operating.

- Methods of collecting habitat and salmonid data should continue to be evaluated in the light of technological advances particularly with regard to exploring the potential benefits of generating in-stream habitat information from remotely sensed data. In this regard it is important that research not only evaluates the accuracy of such techniques but also their practicality, if they are to become beneficial tools for fisheries managers.

Appendix 1: Fisheries Data

The data which follows has been collected by and remains the property of Eden Rivers Trust (Registered Charity No. 1059534). The data may not be used, copied, disclosed, published, sold, assigned, leased, sub-licensed, marketed or transferred without the prior permission and acknowledgment of Eden Rivers Trust. Whilst Eden Rivers Trust has made every effort to ensure the accuracy of the data at the time of survey Eden Rivers Trust is not responsible for any changes that may have taken place since the date of the survey.

Year of survey	Survey Type	Fisheries Officer
2002	Semi-quantitative electrofishing	Alistair Maltby
2003	Semi-quantitative electrofishing	Sara Townsend-Cartwright
2004	Semi-quantitative electrofishing	Judith Brown
2005 Fry	Semi-quantitative electrofishing	Judith Brown
2005 Parr	Quantitative cluster electrofishing Site names beginning with Q represent triple shock catch depletion sites. Those beginning with C represent single pass stop netted sites.	Judith Brown
2006 Trout fry	Semi-quantitative electrofishing	Judith Brown

2002 Survey: Semi-quantitative electrofishing (Eden Rivers Trust)

			No, of fish caught in 5 minutes of fishing					
Date of survey	Sub-catchment	Grid Reference	0+ trout	0+ salmon	Total 0+ salmon & trout	1+ and older trout	1+ and older salmon	No of 0+ salmon and trout missed
12.8.02	Aira Beck	NY3710020800	0	0	0	2	0	1
12.8.02	Aira Beck	NY3993821185	0	0	0	0	0	0
13.8.02	Aira Beck	NY3940221514	0	0	0	1	0	0
12.8.02	Aira Beck	NY4022819879	1	11	12	0	1	3
12.8.02	Aira Beck	NY3815921346	1	0	1	2	0	1
12.8.02	Aira Beck	NY3821021522	2	0	2	0	0	1
14.8.02	Aira Beck	NY4000020400	5	0	5	3	0	3
1.10.02	Argill Beck	NY8255812914	0	0	0	4	0	0
2.10.02	Augill Beck	NY8157414690	1	0	1	5	0	2
2.10.02	Augill Beck	NY7956314032	1	0	1	6	2	1
1.10.02	Belah	NY8350010800	0	0	0	3	0	2
1.10.02	Belah	NY8472809434	0	0	0	5	0	0
18.7.02	Belah	NY7839211940	0	26	26	0	0	6
18.7.02	Belah	NY7774512559	0	22	22	1	0	6
18.7.02	Belah	NY7952612124	0	15	15	0	0	6
1.10.02	Belah	NY8237012243	1	5	6	1	2	7
18.7.02	Belah	NY8165512128	2	8	10	0	5	6
14.8.02	Boredale	NY4267419076	34	0	34	0	0	5
14.8.02	Boredale	NY4157616323	49	0	49	2	1	6
14.8.02	Boredale	NY4196617127	67	0	67	0	0	6
14.8.02	Boredale	NY4239618150	78	0	78	0	0	4
14.8.02	Boredale	NY4201117148	90	0	90	0	0	9
24.7.02	Briggle	NY5648035133	0	3	3	0	1	2
24.7.02	Briggle	NY5767434627	0	0	0	0	0	0
8.8.02	Briggle	NY6054532864	2	0	2	0	0	0
8.8.02	Briggle	NY5883833650	3	0	3	0	0	0
8.8.02	Briggle	NY6153432601	10	3	13	0	0	5
8.8.02	Briggle	NY6295832296	14	0	14	0	0	6
8.8.02	Briggle	NY6469332512	21	0	21	8	0	4
13.9.02	Cairn Beck	NY5226451692	1	0	1	4	0	3
13.9.02	Cairn Beck	NY5148253229	1	0	1	3	0	2
13.9.02	Cairn Beck	NY5036854406	2	0	2	6	0	3
13.9.02	Cairn Beck	NY5589349124	15	0	15	2	0	8
4.9.02	Caldbeck	NY3406639780	0	28	28	1	1	6
4.9.02	Caldbeck	NY3313039851	2	37	39	0	1	4
3.9.02	Caldew	NY3495135035	0	2	2	1	3	2
3.9.02	Caldew	NY3347632559	0	12	12	2	9	3
3.9.02	Caldew	NY3352134977	0	0	0	5	0	0
3.9.02	Caldew	NY3264632735	0	0	0	12	0	2
30.8.02	Caldew	NY3014030730	0	17	17	0	0	7
4.9.02	Caldew	NY3430838893	0	3	3	0	3	3
4.9.02	Caldew	NY3431239856	0	33	33	1	6	6
5.9.02	Caldew	NY3667744483	0	22	22	0	3	0
6.9.02	Caldew	NY3696148895	0	7	7	0	1	10
6.9.02	Caldew	NY3952753886	0	6	6	0	1	2
6.9.02	Caldew	NY3724345522	0	7	7	0	1	12
6.9.02	Caldew	NY3779350941	0	5	5	0	1	8
29.8.02	Caldew	NY3246332417	1	8	9	0	0	5
3.9.02	Caldew	NY3617537726	1	20	21	0	1	6
3.9.02	Caldew	NY3654332658	2	20	22	0	3	4
3.9.02	Caldew	NY3506538214	2	9	11	1	2	4
3.9.02	Caldew	NY3476331916	2	12	14	0	4	0
3.9.02	Caldew	NY3634133400	2	27	29	0	5	4
3.9.02	Caldew	NY3578131988	2	16	18	0	8	4
3.9.02	Caldew	NY3619034911	2	32	34	0	0	3

2002 Survey: Semi-quantitative electrofishing (Eden Rivers Trust)

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4.9.02	Caldew	NY3586442251	2	6	8	0	0	4
6.9.02	Caldew	NY3783346872	2	2	4	1	2	5
4.9.02	Caldew	NY3162331532	3	14	17	0	1	4
3.9.02	Caldew	NY3559435390	4	48	52	1	6	6
30.8.02	Caldew	NY3139031406	6	0	6	0	0	6
24.9.02	Cambeck	NY5260067700	0	0	0	1	0	0
24.9.02	Cambeck	NY5120064300	1	0	1	5	0	0
24.9.02	Cambeck	NY5345168929	1	0	1	0	0	0
24.9.02	Cambeck	NY5090063000	1	3	4	1	0	2
24.9.02	Cambeck	NY5110063500	8	4	12	0	0	4
1.10.02	Coldkeld Beck	NY8250010200	3	0	3	12	0	12
3.10.02	Croglin	NY5753047023	0	0	0	7	0	0
3.10.02	Croglin	NY5512845134	1	0	1	3	0	1
12.07.02	Crowdundale	NY6394629078	0	30	30	0	1	4
12.7.02	Crowdundale	NY6115528161	0	33	33	0	0	6
15.7.02	Crowdundale	NY6378730105	0	40	40	1	3	3
15.7.02	Crowdundale	NY6448830499	0	6	6	0	4	4
15.7.02	Crowdundale	NY6569531371	0	13	13	3	8	4
12.7.02	Crowdundale	NY6560228560	1	37	38	2	2	8
12.7.02	Crowdundale	NY6450828894	1	22	23	0	5	4
12.7.02	Crowdundale	NY6186428266	2	40	42	0	0	5
12.7.02	Crowdundale	NY6274628712	2	30	32	0	1	0
15.7.02	Crowdundale	NY6303629301	2	19	21	1	1	2
15.7.02	Crowdundale	NY6660631343	2	6	8	1	8	4
15.7.02	Crowdundale	NY6595831632	3	0	3	3	0	2
12.7.02	Crowdundale	NY6641828430	7	0	7	0	2	4
12.7.02	Crowdundale	NY6782829032	10	0	10	6	0	2
22.8.02	Dacre	NY4534326084	0	25	25	0	1	4
22.8.02	Dacre	NY4390426493	0	30	30	1	3	5
22.8.02	Dacre	NY4757126861	0	12	12	1	2	2
22.8.02	Dacre	NY4329826728	0	42	42	2	6	6
22.8.02	Dacre	NY4648826249	0	17	17	0	0	5
22.8.02	Dacre	NY4247826290	1	57	58	1	12	8
22.8.02	Dacre	NY4437325630	1	0	1	1	0	0
20.8.02	Deepdale	NY3967214088	0	0	0	0	0	0
20.8.02	Deepdale	NY4007314550	3	1	4	1	4	1
20.8.02	Deepdale	NY3916813498	3	0	3	0	0	1
10.8.02	Deepdale	NY3916813572	4	0	4	1	0	1
27.9.02	Dovedale Beck	NY3915011476	8	3	11	2	1	14
1.10.02	Eamont	NY5601829133	0	1	1	0	1	0
1.10.02	Eamont	NY5074428223	0	25	25	0	4	10
20.8.02	Eamont	NY4749925726	0	1	1	0	0	0
20.8.02	Eamont	NY4953027693	0	18	18	0	0	4
20.8.02	Eamont	NY4810926630	0	2	2	0	0	0
24.9.02	Eamont	NY5720030400	0	4	4	0	0	20
24.9.02	Eamont	NY5257628746	0	15	15	0	15	13
17.9.02	Eden	NY7746409382	0	0	0	0	0	0
18.9.02	Eden	NY7726506219	0	0	0	1	0	0
18.9.02	Eden	NY7657111460	0	15	15	0	0	5
19.9.02	Eden	NY7686412118	0	8	8	0	0	4
19.9.02	Eden	NY7580513634	0	11	11	0	0	6
19.9.02	Eden	NY7805604391	0	0	0	0	0	0
19.9.02	Eden	NY7710010445	0	27	27	0	0	2
19.9.02	Eden	NY7422515139	0	18	18	0	0	6
18.9.02	Eden	SD7761097849	1	0	1	3	0	0

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18.9.02	Eden	SD7760097900	1	0	1	4	0	1
18.9.02	Eden	NY7820002700	1	0	1	0	0	0
18.9.02	Eden	SD7781296616	1	0	1	0	0	1
19.9.02	Eden	NY7699413237	1	19	20	0	0	6
18.9.02	Eden Hellgill	NY7788496545	2	0	2	3	0	0
25.7.02	Gelt	NY5753053739	0	0	0	3	1	0
25.7.02	Gelt	NY5618855614	0	0	0	3	1	0
26.7.02	Gelt	NY5794251343	0	0	0	1	1	0
26.7.02	Gelt	NY5858951157	0	4	4	0	2	2
26.7.02	Gelt	NY5764452435	0	1	1	1	4	2
26.7.02	Gelt	NY5752353285	0	3	3	2	5	0
19.7.02	Gelt	NY4995059438	1	9	10	0	4	0
25.7.02	Gelt	NY5841253220	1	0	1	7	0	0
25.7.02	Gelt	NY5742053869	1	1	2	1	2	0
25.7.02	Gelt	NY5709754297	2	0	2	0	0	0
26.7.02	Gelt	NY5310057524	2	17	19	1	6	2
26.7.02	Gelt	NY5245558814	5	6	11	1	7	3
9.7.02	Gelt	NY5137659366	5	1	6	1	6	1
9.7.02	Gelt	NY5365256560	5	16	21	1	7	4
19.7.02	Gelt	NY5481356221	7	0	7	1	2	0
19.7.02	Gelt	NY5057159426	7	11	18	0	5	3
9.7.02	Gelt	NY5368556495	26	6	32	2	0	0
20.8.02	Glencoyne	NY3858918733	1	0	1	0	0	1
20.8.02	Glencoyne	NY3792418628	9	0	9	1	0	2
19.8.02	Glenridding	NY3736017169	0	0	0	0	0	0
19.8.02	Glenridding	NY3674417388	0	0	0	0	0	0
19.8.02	Glenridding	NY3872816926	4	0	4	0	0	3
27.9.02	Glenridding Beck	NY3737117208	2	1	3	4	0	1
19.8.02	Goldrill	NY4028213616	0	3	3	0	0	2
19.8.02	Goldrill	NY3944916415	0	1	1	0	0	0
19.8.02	Goldrill	NY4047414606	2	3	5	0	0	3
	Grisedale	NY3929516252	0	1	1	0	0	0
	Grisedale	NY3640014600	0	0	0	0	0	0
	Grisedale	NY3763315299	0	0	0	1	0	0
	Grisedale	NY3892215988	0	0	0	0	2	2
	Grisedale	NY3824115770	1	0	1	2	0	1
24.6.02	Grisedale	NY6824115874	2	0	2	3	1	0
	Grisedale	NY3702815092	2	0	2	0	0	0
17.9.02	Hartley Beck	NY7811109035	43	0	43	11	0	5
26.9.02	Hayeswater Beck	NY4200613037	0	0	0	2	0	2
26.9.03	Hayeswater Beck	NY4156612725	0	0	0	3	9	5
26.9.04	Hayeswater Beck	NY4185212095	0	0	0	4	0	2
27.9.02	Hayeswater Beck	NY3998111935	1	8	9	0	2	2
26.9.02	Hayeswater Beck	NY4057713167	4	2	6	0	3	2
26.9.02	Hayeswater Beck	NY4113212880	5	0	5	0	2	7
22.7.02	Helm	NY7099313795	0	16	16	0	0	4
22.7.02	Helm	NY7097614810	0	5	5	0	1	0
22.7.02	Helm	NY7029316738	0	14	14	0	1	3
23.7.02	Helm	NY7130012000	0	0	0	0	0	0
24.7.02	Helm	NY7091909996	3	0	3	1	1	2
24.7.02	Helm	NY7055409112	3	0	3	0	0	3
24.7.02	Helm	NY7045808362	7	0	7	0	0	2
24.7.02	Helm	NY7140811064	9	0	9	0	0	3
27.9.02	Heltondale Beck	NY4928320254	0	0	0	5	2	0
27.9.02	Heltondale Beck	NY4829319653	0	18	18	4	5	9

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27.9.02	Heltondale Beck	NY5049920732	0	7	7	0	6	4
8.7.02	Hilton Beck	NY7089419385	0	5	5	0	0	0
8.7.02	Hilton Beck	NY7076018799	0	5	5	0	0	1
8.7.02	Hilton Beck	NY7074919990	0	3	3	0	1	2
8.7.02	Hilton Beck	NY7356220854	0	8	8	0	2	1
8.7.02	Hilton Beck	NY7255120498	0	7	7	0	3	1
8.7.02	Hilton Beck	NY7166320389	0	11	11	0	4	2
8.7.02	Hilton Beck	NY7420821055	0	6	6	2	6	3
8.7.02	Hilton Beck	NY7621622585	1	0	1	2	0	0
8.7.02	Hilton Beck	NY7454821023	1	5	6	0	6	0
8.7.02	Hilton Beck	NY7564221930	1	13	14	1	7	2
18.7.02	Hoff Beck	NY6647520984	0	13	13	1	0	0
24.6.02	Hoff Beck	NY6724018449	0	5	5	0	0	0
24.6.02	Hoff Beck	NY6709619147	0	0	0	0	0	0
24.6.02	Hoff Beck	NY6717618921	0	0	0	0	0	0
24.6.02	Hoff Beck	NY6716518587	0	1	1	0	0	0
24.6.02	Hoff Beck	NY6742218017	0	6	6	0	0	0
25.06.02	Hoff Beck	NY6846614921	0	0	0	0	0	1
25.6.02	Hoff Beck	NY6623913962	0	0	0	0	0	0
25.6.02	Hoff Beck	NY6682614313	0	0	0	0	0	0
25.6.02	Hoff Beck	NY6661914175	0	0	0	2	0	0
25.6.02	Hoff Beck	NY6560413751	0	0	0	2	0	0
25.6.02	Hoff Beck	NY6819115539	0	0	0	0	0	0
25.6.02	Hoff Beck	NY6759216917	0	0	0	0	0	0
25.6.02	Hoff Beck	NY6714814368	0	0	0	0	0	0
18.7.02	Hoff Beck	NY6664520451	1	3	4	0	1	0
25.6.02	Hoff Beck	NY6785416252	1	0	1	1	0	1
25.06.02	Hoff Beck	NY6813615372	2	0	2	4	0	0
	Hoff Beck	NY6689519656	3	0	3	3	0	0
25.6.02	Hoff Beck	NY6763316638	5	0	5	0	0	0
25.6.02	Hoff Beck	NY6821916052	9	0	9	1	1	6
1.10.02	Howgill Syke	NY8259510255	7	0	7	15	0	5
21.8.02	Irthing	NY6327566495	0	13	13	0	6	3
21.8.02	Irthing	NY6347267788	0	7	7	0	5	2
23.8.02	Irthing	NY5813164222	0	10	10	0	1	2
23.8.02	Irthing	NY6229466590	0	4	4	1	3	5
23.8.02	Irthing	NY6043965116	0	5	5	0	0	0
23.8.02	Irthing	NY6219066166	0	25	25	0	1	4
27.8.02	Irthing	NY5117662339	0	21	21	0	0	0
27.8.02	Irthing	NY5404663310	0	20	20	0	0	2
27.8.02	Irthing	NY5644363953	0	18	18	0	2	4
27.8.02	Irthing	NY5356163461	0	6	6	0	0	0
28.8.02	Irthing	NY4925959824	0	9	9	0	1	0
28.8.02	Irthing	NY4924660280	0	17	17	0	2	3
28.8.02	Irthing	NY5044261078	0	4	4	0	0	3
29.8.02	Irthing	NY4858658056	0	4	4	0	0	3
3.10.02	Irthing	NY6815274302	0	0	0	3	0	0
3.10.02	Irthing	NY6611776596	0	0	0	0	0	0
21.8.02	Irthing	NY6343468251	1	8	9	0	4	0
23.8.02	Irthing	NY6148465956	1	12	13	0	5	2
27.8.02	Irthing	NY5572563447	1	9	10	0	1	4
27.8.02	Irthing	NY5512963353	1	22	23	0	0	6
23.7.02	Irthing	NY6415573585	4	0	4	0	0	0
30.7.02	Kingwater	NY5967768544	0	0	0	2	0	0
30.7.02	Kingwater	NY5270764420	0	6	6	1	0	3

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30.7.02	Kingwater	NY5367965486	0	17	17	0	0	5
30.7.02	Kingwater	NY5643367249	2	3	5	0	0	1
30.7.02	Kingwater	NY5762267456	2	2	4	0	0	3
30.7.02	Kingwater	NY5550066769	2	10	12	0	0	0
30.7.02	Kingwater	NY5862568078	3	3	6	0	0	0
30.7.02	Kingwater	NY5501166474	4	3	7	0	2	3
23.7.02	Kingwater	NY6132972484	5	0	5	1	0	0
8.8.02	Kirkland	NY6547032858	7	0	7	4	0	4
27.9.02	Kirkstone Beck	NY4025110289	0	0	0	9	0	0
27.9.02	Kirkstone Beck	NY4008612469	1	1	2	1	1	4
27.9.02	Kirkstone Beck	NY4030109842	1	0	1	7	0	0
27.9.02	Kirkstone Beck	NY3997710814	5	2	7	5	0	9
27.9.02	Kirkstone Beck	NY3993710862	6	6	12	1	0	15
31.7.01	Leith	NY5517821842	0	0	0	0	0	0
31.7.02	Leith	NY5872024491	0	2	2	0	2	0
31.7.02	Leith	NY5622125022	0	3	3	1	2	1
31.7.02	Leith	NY5551724765	0	0	0	0	0	0
31.7.02	Leith	NY5768324725	0	22	22	0	0	5
31.7.02	Leith	NY5507524010	0	0	0	1	0	0
	Leith	NY6009024518	0	0	0	0	0	0
	Leith	NY5881624331	0	0	0	0	0	0
	Leith	NY5506023875	0	0	0	0	0	2
	Leith	NY5550724682	0	0	0	0	0	0
	Leith	NY5515721686	0	0	0	0	0	1
	Leith	NY5574520408	0	0	0	0	0	1
	Leith	NY5512822759	0	0	0	0	0	1
	Leith	NY5615424950	0	0	0	0	1	0
31.7.02	Leith	NY5511322651	1	0	1	0	0	0
31.7.02	Leith	NY5950724462	1	6	7	0	0	3
	Leith	NY5767124761	1	0	1	0	0	0
	Leith	NY5706625187	1	0	1	0	0	0
	Leith	NY5587519519	2	0	2	1	0	0
	Leith	NY5551717837	3	0	3	0	0	0
1.8.02	Lowther	NY5174021373	0	16	16	0	0	4
1.8.02	Lowther	NY5194423355	0	17	17	0	0	6
1.8.02	Lowther	NY5220225450	0	11	11	0	1	4
1.8.02	Lowther	NY5276426113	0	3	3	1	0	3
22.8.02	Lowther	NY5267328650	0	8	8	0	1	2
25.7.02	Lowther	NY5206817990	0	28	28	0	1	0
25.7/02	Lowther	NY5269228652	0	2	2	0	1	0
5.8.02	Lowther	NY5278117548	0	9	9	0	4	3
6.8.02	Lowther	NY5116816110	0	1	1	0	0	3
6.8.02	Lowther	NY5344916429	0	3	3	0	0	2
6.8.02	Lowther	NY5473215353	0	25	25	0	4	4
6.8.02	Lowther	NY5119516052	0	17	17	1	0	3
7.8.02	Lowther	NY5482914039	0	1	1	0	1	1
7.8.02	Lowther	NY5550212046	0	14	14	0	4	3
1.8.02	Lowther	NY5164522115	1	13	14	0	0	4
5.8.02	Lowther	NY5152019764	1	8	9	0	0	3
5.8.02	Lowther	NY5169818379	1	21	22	0	0	4
6.8.02	Lowther	NY5095315170	1	0	1	0	0	0
6.8.02	Lowther	NY5597812659	1	37	38	0	1	5
6.8.02	Lowther	NY5117716055	1	3	4	0	0	0
6.8.02	Lowther	NY5505914336	1	44	45	0	1	7
5.8.02	Lowther	NY5177021403	2	10	12	0	0	2

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5.8.02	Lowther	NY5179017872	2	9	11	0	1	4
6.8.02	Lowther	NY5045714375	2	0	2	2	0	2
5.8.02	Lowther	NY5180218115	3	8	11	0	1	0
7.8.02	Lowther	NY5526213479	3	3	6	0	0	2
6.8.02	Lowther	NY5200216607	4	12	16	0	2	0
6.8.02	Lowther	NY5177717702	8	19	27	1	0	4
11.07.02	Lyvennet	NY6144221935	0	0	0	0	0	0
11.07.02	Lyvennet	NY6218518140	0	12	12	1	0	6
11.07.02	Lyvennet	NY6178720947	0	2	2	0	0	2
11.07.02	Lyvennet	NY6204519015	0	6	6	0	0	2
11.07.02	Lyvennet	NY6223915215	0	0	0	1	0	1
11.7.02	Lyvennet	NY6197620158	0	0	0	1	0	0
11.7.02	Lyvennet	NY6122823269	0	10	10	0	0	5
11.7.02	Lyvennet	NY6210020200	0	0	0	0	0	0
11.7.02	Lyvennet	NY6064823880	0	1	1	0	0	1
11.7.02	Lyvennet	NY6017524531	0	0	0	0	0	0
11.7.02	Lyvennet	NY6062024030	0	0	0	1	0	0
12.07.02	Lyvennet	NY6052025097	0	14	14	0	0	3
12.7.02	Lyvennet	NY6109725118	0	6	6	0	0	2
11.7.02	Lyvennet	NY6007422248	1	0	1	1	0	0
3.10.02	Petteril	NY4751340778	0	0	0	5	0	0
3.10.02	Petteril	NY4635342694	0	0	0	3	0	0
3.10.02	Petteril	NY4905938849	0	0	0	1	0	0
	Petteril	NY4734441747	0	0	0	0	0	0
	Petteril	NY4836539661	0	0	0	0	0	0
	Petteril	NY4750440711	0	0	0	0	0	0
	Petteril	NY4630242697	0	0	0	0	0	0
16.8.02	Rampsgill	NY4385916362	10	0	10	1	3	6
16.8.02	Rampsgill	NY4398414441	12	0	12	6	0	4
16.8.02	Rampsgill	NY4403215436	25	0	25	4	0	7
18.9.02	Sandwath Sike	NY7690310431	1	7	8	0	0	7
14.8.02	Sandwick	NY4235919715	2	2	4	1	5	0
15.8.02	Sandwick	NY4316719116	8	17	25	0	0	6
16.8.02	Sandwick	NY4374217574	8	9	17	0	3	6
16.8.02	Sandwick	NY4355816807	12	5	17	0	2	7
16.8.02	Sandwick	NY4336218274	12	15	27	1	0	8
16.8.02	Sandwick	NY4361817514	17	8	25	0	2	6
16.8.02	Sandwick	NY4297315963	32	0	32	0	1	3
16.8.02	Sandwick	NY4273215285	37	1	38	0	0	7
16.7.02	Scandal Beck	NY7436709968	0	13	13	0	1	5
16.7.02	Scandal Beck	NY7234306103	0	3	3	2	0	2
16.7.02	Scandal Beck	NY7197504672	0	0	0	2	3	0
16.7.02	Scandal Beck	NY7193905254	0	2	2	0	0	0
16.7.02	Scandal Beck	NY7271307023	0	13	13	2	4	7
16.7.02	Scandal Beck	NY7315508129	1	7	8	0	3	9
16.7.02	Scandal Beck	NY7410109415	3	9	12	0	0	4
16.7.02	Scandal Beck	NY7244203977	4	0	4	2	0	3
16.7.02	Scandal Beck	NY7481310840	4	36	40	0	1	4
7.8.02	Swindale	NY5275015028	0	23	23	0	3	3
7.8.02	Swindale	NY5352515963	0	9	9	0	5	3
7.8.02	Swindale	NY5205013515	1	3	4	0	3	2
7.8.02	Swindale	NY5157813266	4	0	4	0	2	2
7.8.02	Swindale	NY5042311926	4	1	5	0	4	3
7.8.02	Swindale	NY5104512737	6	24	30	0	1	3
1.10.02	Swindale Beck	NY7774413721	0	12	12	1	6	12

2002 Survey: Semi-quantitative electrofishing (Eden Rivers Trust)

			No, of fish caught in 5 minutes of fishing					
Date of survey	Sub-catchment	Grid Reference	0+ trout	0+ salmon	Total 0+ salmon & trout	1+ and older trout	1+ and older salmon	No of 0+ salmon and trout missed
1.10.02	Swindale Beck	NY7957714642	0	0	0	12	1	0
19.9.02	Swindale Beck	NY7699413237	0	12	12	0	0	4
1.10.02	Swindale Beck	NY8180318157	1	0	1	5	0	1
10.7.02	Trout Beck	NY6396025444	0	4	4	0	0	0
10.7.02	Trout Beck	NY6464724480	0	17	17	0	0	4
10.7.02	Trout Beck	NY6588524056	0	5	5	0	0	0
10.7.02	Trout Beck	NY6749024182	0	15	15	0	0	4
10.7.02	Trout Beck	NY6902228251	0	0	0	1	0	2
10.7.02	Trout Beck	NY6692524442	0	8	8	0	1	0
10.7.02	Trout Beck	NY6848725119	0	0	0	8	1	0
8.7.02	Trout Beck	NY6980023200	0	1	1	1	6	0
10.7.02	Trout Beck	NY6765324209	1	40	41	1	2	0
10.7.02	Trout Beck	NY6856125486	1	2	3	2	4	0
11.07.02	Trout Beck	NY6200414734	1	0	1	0	0	0
8.7.02	Trout Beck	NY7027522962	1	6	7	0	10	2
12.7.02	Trout Beck	NY6837826530	4	26	30	1	2	4
10.7.02	Trout Beck	NY6781823861	6	3	9	0	0	2
12.7.02	Trout Beck	NY6849127134	7	18	25	6	15	3
12.7.02	Trout Beck	NY6837924977	8	1	9	4	2	0
10.7.02	Trout Beck	NY6890023307	14	13	27	0	1	4
1.10.02	Well Head Syke	NY8080016000	0	0	0	9	0	2

2003 Survey: Semi-quantitative electrofishing (Eden Rivers Trust)

				No. of fish caught in 5 minutes of fishing					
Date of survey	Sub-catchment	Easting	Northing	0+ trout	0+ salmon	Total 0+ salmon & trout	1+ trout	1+ salmon	No. of 0+ salmon & trout missed
6.8.03	Aira	337275	520725	1	0	0	3		0
6.8.03	Aira	336312	520925	0	0	0	1		0
6.8.03	Aira	338278	521467	1	0	0	1		0
6.8.03	Aira	340179	519810	0	2	2		2	4
6.8.03	Aira	340098	520301	17	15	1	1	1	15+
6.8.03	Aira	339793	521447	0	0	0	1		1
26.8.03	Argill	368752	513900	3	0	0	4		2
26.8.03	Argill	381620	514682	11	0	0	13		
11.8.03	Asby Beck	380447	514344	4	0	0	2		4
24.7.03	Augill	382448	512703	1	19	0	3		25+
24.7.03	Augill	385178	513568	12	8	0			25+
24.8.03	Belah	382316	512006	26	69	7	1	7	
24.8.03	Belah	383212	511207	17	0	0	4		5
24.8.03	Belah	383738	519679	18	0	0	9		5
24.8.03	Belah	384824	509443	26	0	0	1		
26.8.03	Belah	381581	512121	7	50	6	1	6	
27.8.03	Briggle Beck	361702	532633	19	0	0	7		6+
5.9.03	Briggle Beck	358790	533480	10	2	1	5	1	
5.9.03	Briggle Beck	357677	534629	5	10	2	3	2	10
12.9.03	Cairn Beck	350452	554342	7	0	0			0
12.9.03	Cairn Beck	351000	551832	3	0	0	1		1
12.9.03	Cairn Beck	353966	550473	2	0	0			1
15.9.03	Cairn Beck	347710	556522	0	47	4	1	4	10
15.9.03	Cairn Beck	348547	555244	8	0	0	7		5
15.9.03	Cairn Beck	349778	554788	10	0	0	1		0
15.9.03	Cairn Beck	351506	553245	6	0	0	1		2
15.9.03	Cairn Beck	344136	556372	0	1	0			0
5.9.03	Caldbeck	333049	539878	9	55	6	2	6	10
26.9.03	Caldbeck	329540	538147	10	0	0	3		5
26.9.03	Caldbeck	330049	535956	32	0	4		4	6
26.9.03	Caldbeck	329950	535155	33	0	0	7		4
26.9.03	Caldbeck	330018	538816	21	0	0	4		5
26.9.03	Caldbeck	329849	537289	10	0	0	4		7
26.9.03	Caldbeck	328734	537048	3	0	0	1		4
1.9.03	Caldew	330140	530774	4	5	2	2	2	4
1.9.03	Caldew	330422	530523	1	13	4		4	5
1.9.03	Caldew	331130	531182	1	2	4		4	4
3.9.03	Caldew	336358	535449	1	20	6	1	6	12
3.9.03	Caldew	335788	538134	0	17	5		5	10
3.9.03	Caldew	330113	539899	25	0	0	10		5
3.9.03	Caldew	336561	532746	0	6	0	1		3
3.9.03	Caldew	336508	532963	1	68	3		3	10
3.9.03	Caldew	336048	534098	4	73	5	1	5	10
3.9.03	Caldew	333429	532502	0	5	4	1	4	
3.9.03	Caldew	332691	532691	2	0	0	3		4
3.9.03	Caldew	335705	532011	0	34	3		3	
8.9.03	Cambeck	353498	568842	2	0	0	1		1
8.9.03	Cambeck	352630	567491	5	0	0	3		4
9.9.03	Cambeck	350966	562543	3	15	1		1	3
9.9.03	Cambeck	352743	566633	0	0	0	2		0

2003 Survey: Semi-quantitative electrofishing (Eden Rivers Trust)

				No. of fish caught in 5 minutes of fishing					
Date of survey	Sub-catchment	Easting	Northing	0+ trout	0+ salmon	Total 0+ salmon & trout	1+ trout	1+ salmon	No. of 0+ salmon & trout missed
9.9.03	Cambeck	351853	565354	2	0	0			0
9.9.03	Castle Beck	355850	563400	55	22	0	2		10
9.9.03	Castle Beck	356282	562658	29	0	0	16		10
7.8.03	Colby Beck	366500	521000	0	40	2	1	2	6
7.8.03	Colby Beck	366772	520234	0	13	1		1	7
5.9.03	Croglin	355146	545171	19	0	0	2		5
5.9.03	Croglin	354492	543795	11	0	0	1		5
5.9.03	Croglin	356282	546527	17	0	0	2		10
5.9.03	Croglin	358088	547264	17	0	0	8		10
5.9.03	Croglin	355871	545909	24	0	0	7		10
21.8.03	Crowdundle	365873	528633	87	6	0	2		50+
21.8.03	Crowdundle	365447	528550	22	63	1	2	1	50+
21.8.03	Crowdundle	364500	528862	12	140	0			
22.8.03	Crowdundle	365189	531019	15	52	11		11	20+
22.8.03	Crowdundle	364807	530789	3	42	2	3	2	10+
22.8.03	Crowdundle	364498	530527	39	47	6		6	25+
22.8.03	Crowdundle	363123	529079	11	124	4		4	50+
22.8.03	Crowdundle	363003	529115	7	33	4		4	8
22.8.03	Crowdundle	362553	528724	1	101	5		5	40-50
11.9.03	Crowdundle	369374	529982	42	0	42	4		8
11.9.03	Crowdundle	369845	530002	6	0	6	13	0	3
10.9.03	Dacre	346298	526295	1	10	5	1	5	5
10.9.03	Dacre	345272	526094	1	14	4		4	5
10.9.03	Dacre	343999	526488	2	52	10	1	10	6
17.9.03	Dacre	343700	526446	1	45	1		1	10
22.8.03	Daleraven	355410	539546	0	43	0			8+
12.9.03	Deepdale	339921	514386	21	5	26	0	3	5
17.7.03	Eden	336936	558682	0	4	0			3
17.7.03	Eden	343692	558549	0	3	0			5
17.7.03	Eden	346736	557927	0	42	2		2	
17.7.03	Eden	346941	556721	0	24	0			20+
17.7.03	Eden	346798	555461	0	21	0			15+
22.7.03	Eden	356405	539494	0	19	1		1	12
22.7.03	Eden	355947	537729	0	14	0			14
22.7.03	Eden	361215	525889	0	33	0			22
23.7.03	Eden	347515	553819	1	7	0			6
23.7.03	Eden	351700	548500	0	23	0			lots
23.7.03	Eden	351362	546903	0	1	0			7
23.7.03	Eden	348324	552270	0	2	0			3
23.7.03	Eden	355913	537390	0	23	0	1		15+
23.7.03	Eden	351213	546480	1	6	0			5+
24.7.03	Eden	355408	534551	0	15	0			25+
24.7.03	Eden	356163	533693	0	11	0			10
14.8.03	Eden	363290	525050	0	17	0			3
18.9.03	Eden	377342	504307	0	0	0			0
18.9.03	Eden	378128	503060	1	0	0	1		0
19.9.03	Eden	378265	499891	0	0	0			0
11.9.03	Eden	377894	508178	0	60	60	0	4	4
11.9.03	Eden	377129	506624	3	0	3	0		2
11.9.03	Eden	377140	507318	2	0	2	0	0	0

2003 Survey: Semi-quantitative electrofishing (Eden Rivers Trust)

				No. of fish caught in 5 minutes of fishing					
Date of survey	Sub-catchment	Easting	Northing	0+ trout	0+ salmon	Total 0+ salmon & trout	1+ trout	1+ salmon	No. of 0+ salmon & trout missed
11.9.03	Eden	377651	509084	1	76	77	0	3	4
14.7.03	Gelt	357539	553326	1	8	3	1	3	6
14.7.03	Gelt	357708	552063	0	15	7		7	5
14.7.03	Gelt	358385	551132	6	5	3		3	4
14.7.03	Gelt	359425	551274	13	0	0	1		2
14.7.03	Gelt	360411	551081	7	0	0	3		0
14.7.03	Gelt	360466	551090	2	0	0			0
14.7.03	Gelt	361338	551109	5	0	0	2		3
14.7.03	Gelt	358462	553274	2	0	0	4		10
14.7.03	Gelt	359103	553306	9	0	0	4		4
22.8.03	Glassonby Beck	358886	539726	24	3	0	1		6+
22.8.03	Glassonby Beck	359386	539677	29	4	0	3		
27.9.03	Glassonby Beck	360965	538800	3	0	0	5		
16.9.03	Glencoyne Beck	336712	518799	9	0	0	1		3
16.9.03	Glencoyne Beck	344941	522579	14	0	1	0	1	0
08.8.03	Glenridding	336140	517131	2	0	0	1		4
08.8.03	Glenridding	338768	516908	12	1	1	6	1	4
08.8.04	Glenridding	338461	516828	6	0	0	2		4
08.8.05	Glenridding	337638	517079	15	0	1	5	1	3
08.8.06	Glenridding	336874	517304	0	0	0			2
08.8.03	Goldrill	339927	515713	6	0	1		1	6
12.0.03	Goldrill	340271	514006	6	31	37			5
12.9.04	Goldrill	341616	512885	1	0	1	1		4
12.9.03	Goldrill Beck	341593	512662	24	0	10	1	10	10
12.9.03	Goldrill Beck	340588	513162	4	16	0			5
15.7.03	Grisedale Beck	338327	515763	14	0	1	4	1	6
16.7.03	Grisedale Beck	337358	515156	7	0	1	1	1	4
16.7.03	Grisedale Beck	336423	514652	11	0	1		1	6
16.7.03	Grisedale Beck	335912	513856	1	0	0	2		0
11.9.03	Hartley	377625	509011	18	12	30	4		1
11.9.03	Hartley	378297	508762	9	0	9	2		5
18.9.03	Haweswater Beck	351810	518072	4	31	1		1	0
12.8.03	Helm	370870	517015	0	23	0	1		5
12.8.03	Helm	370274	510703	0	14	0			2
12.8.03	Helm	370668	515720	0	1	0			0
12.8.03	Helm	370703	514333	0	1	0			0
12.8.03	Helm	371194	513654	2	2	0	7		0
12.8.03	Helm	371321	512237	1	1	0	11		0
12.8.03	Helm	371139	510566	25	0	0	1		4
13.8.03	Helm	371315	510862	27	0	0			4
24.9.03	Helm	370956	514916	0	0	0			
17.9.03	Heltondale	348596	519737	1	13	2		2	4
13.8.03	Hilton	371048	518978	0	17	2		2	8
13.8.03	Hilton	372004	519143	1	0	0			0
13.8.03	Hilton	371261	520086	7	100	9		9	20
14.8.03	Hilton	373249	520695	18	56	12	1	12	0
14.8.03	Hilton	373475	520853	5	44	20		20	8
24.8.03	Hilton Beck	376212	522567	4	0	1	4	1	8
24.8.03	Hilton Beck	375314	521493	13	28	6	1	6	30
7.8.03	Hoff Beck	367179	518944	0	26	1		1	8

2003 Survey: Semi-quantitative electrofishing (Eden Rivers Trust)

				No. of fish caught in 5 minutes of fishing					
Date of survey	Sub-catchment	Easting	Northing	0+ trout	0+ salmon	Total 0+ salmon & trout	1+ trout	1+ salmon	No. of 0+ salmon & trout missed
7.8.03	Hoff Beck	367504	517680	0	27	6		6	8
7.8.03	Hoff Beck	367530	516914	0	17	1		1	5
7.8.03	Hoff Beck	368239	515874	1	8	2		2	3
24.9.03	Howe	350374	517657	24	0	0	4		5
24.9.03	Howe	351617	518169	27	12	4	5	4	20+
28.9.03	Howe	347647	517732	18	0	0	11		2
28.9.03	Howe	348672	517916	22	0	0			3
28.9.03	Howe	349680	517760	19	0	0	9		
10.9.03	Howgrain Beck	343216	519082	32	35	8	1	8	15
8.9.03	Irthing	363730	567796	0	8	7	1	7	4
8.9.03	Irthing	367596	574581	2	0	0	9		0
8.9.03	Irthing	368091	574325	0	0	0	5		0
8.9.03	Irthing	366361	570977	2	0	0	2		0
9.9.03	Irthing	355184	563334	0	46	4	3	4	10
2.9.03	Kingwater	352887	564407	1	15	1		1	4
2.9.03	Kingwater	353178	564893	1	12	3	1	3	2
2.9.03	Kingwater	354903	566469	2	10	1	1	1	3
2.9.03	Kingwater	355408	566799	4	14	4		4	4
2.9.03	Kingwater	356802	567329	18	0	1		1	3
2.9.03	Kingwater	357637	567464	14	0	0			3
2.9.03	Kingwater	359837	568594	3	0	0	1		1
2.9.03	Kingwater	360259	569166	2	0	6	3	6	3
2.9.03	Kingwater	361812	570023	0	0	1	7	1	0
10.9.03	Kirkstone Beck	339926	511039	25	9	7	3	7	10
12.9.03	Kirkstone Beck	340016	511985	1	1	2		2	5
16.9.03	Kirkstone Beck	339984	511930	8	0	4		4	3
11.9.03	Ladthwaite	377970	508185	2	5	7	5	3	1
27.7.03	Leith	355184	524076	5	26	0			10
27.7.03	Leith	355137	522976	4	13	0	5		10
29.7.03	Leith	355194	521701	9	3	1		1	4
5.8.03	Leith	359012	524484	0	7	0	2		4
5.8.03	Leith	359970	524295	1	11	0			0
5.8.03	Leith	355539	518065	5	0	0			6
5.8.03	Leith	355744	525079	3	1	0			4
5.8.03	Leith	357099	525140	1	12	0			6
17.9.03	Lowther	355536	512113	1	11	1		1	5
18.9.03	Lowther	353282	516692	0	16	3		3	3
18.9.03	Lowther	352566	517561	0	29	4	1	4	3
19.9.03	Lowther	355992	513744	4	63	3		3	25
19.9.03	Lowther	355163	514296	0	69	0			20
19.9.03	Lowther	355992	512607	0	22	2		2	30
15.7.03	Lyvennet	362123	512694	0	9	0			12
15.7.03	Lyvennet	362171	512801	22	3	0			12
15.7.03	Lyvennet	362261	513862	20	0	0	1		21
15.7.03	Lyvennet	362246	514953	18	0	0			8
15.7.03	Lyvennet	362642	516034	18	2	0	1		8
21.7.03	Lyvennet	362216	518106	2	3	0			2
21.7.03	Lyvennet	362096	518904	0	3	0			5
21.7.03	Lyvennet	361936	570081	0	6	0	1		
21.7.03	Lyvennet	361857	521066	6	3	1	3	1	2

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				No. of fish caught in 5 minutes of fishing					
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21.7.03	Lyvennet	361460	521923	1	0	0			
21.7.03	Lyvennet	361261	523290	2	12	0			
21.7.03	Lyvennet	360035	524644	0	2	0			
22.7.03	Lyvennet	366967	524830	0	35	1		1	8
22.7.03	Lyvennet	360856	525897	0	44	1		1	12
24.9.03	Lyvennet	361869	521132	0	1	1	4	1	2
9.9.03	Milton Beck	354786	561139	14	0	0	10		4
9.9.03	Milton Beck	355315	560611	17	0	0	11		3
24.9.03	Morland Beck	360607	523782	0	4	7		7	
24.9.03	Morland Beck	359977	521284	4	0	0			
24.9.03	Morland Beck	359965	522425	3	0	0	3		
24.9.03	Morland Beck	359875	523472	0	0	0			
24.9.03	Newby Beck	359221	520648	7	0	0	2		
10.9.03	Petteril	345363	532732	9	0	0	8		6
10.9.03	Petteril	346569	532881	10	0	0	4		4
10.9.03	Petteril	348081	531961	3	0	0	10		6
10.9.03	Petteril	349602	533361	0	0	0	5		6+
10.9.03	Petteril	345004	532037	7	0	0	6		8
28.8.03	Petterill	348833	538971	0	0	0			0
28.8.03	Petterill	348020	539981	0	0	0	2		0
28.8.03	Petterill	347521	540941	0	0	0	3		0
28.8.03	Petterill	347250	541932	0	0	0	4		0
29.8.03	Petterill	347521	540941	0	0	0	3		0
29.8.03	Petterill	347250	541932	0	0	0	4		0
29.8.03	Petterill	346299	542700	0	0	0	7		0
29.8.03	Petterill	346731	542605	0	0	0	8		0
29.8.03	Petterill	345047	545230	0	0	0	1		0
29.8.03	Petterill	345009	546390	0	0	0	3		0
29.8.03	Petterill	345201	547479	0	0	0			0
29.8.03	Petterill	344203	549216	0	0	0	3		0
29.8.03	Petterill	349636	535119	0	0	0	4		0
9.9.03	Quarry Beck	355315	563310	6	12	3	3	3	2
9.9.03	Quarry Beck	354607	562201	0	0	0			0
27.7.03	Raven Beck	355720	541079	0	53	1		1	2
27.7.03	Raven Beck	360594	543269	35	0	0	1		0
27.7.03	Raven Beck	357743	541751	9	0	0	8		0
27.7.03	Raven Beck	356721	541635	15	0	0	1		0
27.7.03	Raven Beck	361390	543605	1	0	0	3		1
27.7.03	Raven Beck	361578	543822	5	0	0	13		3
27.7.03	Raven Beck	361477	543449	6	0	0	11		4
27.8.03	Robbery Water	359845	537329	20	1	0			10
27.8.03	Robbery Water	359855	537329	35	10	0	2		10
27.8.03	Robbery Water	356823	535944	1	45	1		1	8+
27.8.03	Robbery Water	359363	536188	16	28	0	3		8+
10.9.03	Sandwick Beck	342618	519170	44	1	3	2	3	5
17.9.03	Sandwick Beck	342433	518217	21	0	0			5
17.9.03	Sandwick Beck	343335	518400	31	5	0	1		
26.8.03	Scale Beck	367238	514530	2	0	0	6		2
24.7.07	Scandal	373575	508580	1	30	0	1		20
29.7.03	Scandal	372568	503873	0	21	2	2	2	3

2003 Survey: Semi-quantitative electrofishing (Eden Rivers Trust)

				No. of fish caught in 5 minutes of fishing					
Date of survey	Sub-catchment	Easting	Northing	0+ trout	0+ salmon	Total 0+ salmon & trout	1+ trout	1+ salmon	No. of 0+ salmon & trout missed
29.7.03	Scandal	372071	505705	2	3	0			3
11.8.03	Scandal	372680	506852	3	22	5	1	5	12
11.8.03	Scandal	374877	510959	1	75	0			15
11.8.03	Scandal	376513	511236	0	20	20		20	6
4.9.03	Swindale	379997	514823	7	0	0	12		10
4.9.03	Swindale	380594	515934	17	0	0	21		10
5.9.03	Swindale	377368	513447	4	51	0			40
5.9.03	Swindale	378784	514175	1	19	0			10
18.9.03	Swindale	350686	512325	4	33	0	1		8
18.9.03	Swindale	351742	513336	0	3	5		5	0
18.9.03	Swindale	352740	515040	0	22	10		10	
18.9.03	Swindale	353554	515957	0	8	7	1	7	0
14.8.03	Troutbeck	363345	525142	1	20	3	1	3	6
14.8.03	Troutbeck	366624	524260	3	63	0	1		0
14.8.03	Troutbeck	367667	524011	0	28	1		1	4
14.8.03	Troutbeck	367630	524185	0	37	3		3	0
14.8.03	Troutbeck	364310	524969	0	28	0			3
14.8.03	Troutbeck	365415	524247	0	16	3		3	8
15.8.03	Troutbeck	368546	525289	16	42	0			40
15.8.03	Troutbeck	368417	526957	20	42	0			25
20.8.03	Troutbeck	368129	523411	2	30	0			20
20.8.03	Troutbeck	370224	522941	7	30	0			25
20.8.03	Troutbeck	369563	523222	11	21	0			16
20.8.03	Troutbeck	372313	523589	26	9	0			20
20.8.03	Troutbeck	368838	527600	7	4	0			50
20.8.03	Troutbeck	367381	528188	13	33	0			35

2004 Survey: Semi-quantitative electrofishing with habitat data (Eden Rivers Trust)

						No. of fish caught in 5 minutes of fishing															
Date of survey	Sub-catchment	Easting	Northing	Altitude (m above sea level)	Temp °C	0+ Trout	0+ Salmon	Total 0+ salmon & trout	1+ and older Trout	1+ and older Salmon	No of 0+ salmon & trout missed	Dominant substrate	Substrate2	Substrate3	Siltation 1=present 0=absent	Bank profile	Erosion type	Stock access 1=yes 0=no	Tunnelling 1=present 0=absent	Fence width (m)	Landuse
16/09/2004	Aira Beck	336942	520747	394	10	2	0	2	4	0	1	Boulders	Cobbles		0	Shallow	Undercut	1	0	0	Unimproved grassland
16/09/2004	Aira Beck	337590	520859	373	11	2	0	2	5	0	1	Cobbles			0	Shallow	Undercut	1	0	0	Unimproved grassland
16/09/2004	Aira Beck	338220	521405	364	11	0	0	0	2	0	0	Cobbles	Pebbles		0	Shallow	None	1	0	10	Unimproved grassland
16/09/2004	Aira Beck	339298	521595	365	11	0	0	0	2	0	0	Cobbles	Boulders	Pebbles	0	Steep	None	0	0	3	Park/garden
29/09/2004	Aira Beck	340204	519954	170	12	5	1	6	1	4	2	Cobbles	Pebbles	Gravel	0	Shallow	None	0	0	10	Improved grassland
14/07/2004	Asby Beck / Hoff	368538	514734	142	19	0	0	0	1	0	1	Pebbles	Cobbles	Gravel	1	Shallow	Earth cliff	1	0	0	Improved grassland
31/08/2004	Bannerdale	343576	516880	207	14	29	8	37	0	0	10	Cobbles	Pebbles	Gravel	0	Shallow	None	0	0	10	Unimproved grassland
23/08/2004	Belah	382425	511929	252	12	0	22	22	1	10	5	Cobbles	Boulders	Pebbles	0	Steep	None	0	0	0	Improved grassland
23/08/2004	Belah	381868	512217	188	12	1	15	16	1	8	6	Cobbles	Boulders	Pebbles	0	Steep	Undercut	0	0	0	Improved grassland
23/08/2004	Belah	380804	512008	188	12	2	20	23	0	6	8	Cobbles	Pebbles	Gravel	0	Shallow	Earth cliff	1	0	0	Improved grassland
23/08/2004	Belah	379304	512077	175	12	0	24	24	0	2	10	Cobbles	Pebbles	Gravel	0	Shallow	None	1	0	0	Improved grassland
23/08/2004	Belah	378385	511952	171	12	0	24	24	0	3	10	Cobbles	Pebbles	Gravel	0	Shallow	None	1	0	0	Improved grassland
24/08/2004	Belah	363133	511241	342	14	0	0	0	7	0	0	Boulders	Cobbles		0	Steep	Earth cliff	0	0	0	Unimproved grassland
23/08/2004	Belah (Argil)	382557	512948	252	12	0	0	0	2	0	0	Bedrock	Cobbles	Pebbles	0	Shallow	Undercut	1	0	7	Improved grassland
31/08/2004	Boredale	342026	517218	198	12	34	0	34	2	0	12	Pebbles	Cobbles	Gravel	0	Shallow	Earth cliff	1	0	0	Unimproved grassland
31/08/2004	Boredale	342525	518400	185	13	17	0	17	0	0	6	Gravel	Pebbles		1	Steep	Undercut	0	0	2	Unimproved grassland
17/08/2004	Briggle	362519	532568	239	16	4	0	4	2	0	2	Sand	Pebbles	Cobbles	1	Shallow	Earth cliff	1	0	0	Improved grassland
17/08/2004	Briggle	358957	533532	236	16	3	0	3	3	0	1	Cobbles	Pebbles	Sand	0	Steep	Undercut	0	1	0	Improved grassland
17/08/2004	Briggle	357915	534437	115	15	4	0	4	0	0	2	Pebbles	Cobbles	Silt/mud	1	Steep	Earth cliff	1	1	0	Improved grassland
17/08/2004	Briggle	356496	535169	98	15	0	11	11	2	1	3	Cobbles	Pebbles	Gravel	0	Steep	None	0	0	2	Improved grassland
17/08/2004	Briggle (Skirwith)	360891	532548	239	17	9	0	9	1	0	3	Cobbles	Pebbles	Gravel	0	Steep	Poached	1	0	1	Improved grassland
27/09/2004	Caldbeck	330162	534667	379	11	4	0	4	8	0	1	Pebbles	Cobbles	Boulders	0	Shallow	None	1	0	0	Unimproved grassland
27/09/2004	Caldbeck	329938	535332	335	12	1	0	1	8	0	0	Cobbles	Boulders	Pebbles	0	Shallow	Undercut	1	0	0	Unimproved grassland
27/09/2004	Caldbeck	330080	536018	302	12	0	0	0	8	0	0	Boulders	Cobbles	Pebbles	0	Shallow	Undercut	1	0	0	Unimproved grassland
27/09/2004	Caldbeck	330106	536718	279	12	0	0	0	2	0	0	Cobbles	Pebbles	Gravel	0	Shallow	Undercut	1	0	0	Unimproved grassland
27/09/2004	Caldbeck	332688	539813	278	13	1	3	4	1	2	2	Cobbles	Pebbles	Gravel	0	Steep	None	0	0	0	Woodland
27/09/2004	Caldbeck	329816	537332	261	12	0	0	0	2	0	0	Pebbles	Gravel		0	Shallow	Poached	1	0	0	Improved grassland
27/09/2004	Caldbeck	329658	537977	237	12	0	0	0	4	0	0	Cobbles	Pebbles	Gravel	0	Shallow	Undercut	1	0	0	Improved grassland
10/09/2004	Caldew	334295	538933	155	14	0	1	1	0	1	0	Boulders			0	Shallow	None	1	1	4	Improved grassland
10/09/2004	Caldew	335178	538192	150	14	0	10	10	0	4	5	Boulders			0	Steep	None	0	0	6	Improved grassland
25/08/2004	Caldew	331365	531342	339	12	2	15	17	0	4	5	Cobbles	Boulders	Pebbles	0	Steep	None	1	0	0	Unimproved grassland
25/08/2004	Caldew	332131	532158	337	13	0	5	5	1	2	2	Cobbles	Boulders	Pebbles	0	Steep	None	1	0	0	Unimproved grassland
10/09/2004	Caldew	338359	537319	189	13	0	16	16	0	8	7	Boulders			0	Shallow	None	1	1	4	Improved grassland
09/09/2004	Caldew	333175	532617	273	11	0	4	4	0	7	3	Boulders	Cobbles		0	Shallow	None	1	0	0	Unimproved grassland
09/09/2004	Caldew	336522	532814	223	12	1	41	42	0	2	3	Cobbles			0	Steep	None	1	0	0	Improved grassland
09/09/2004	Caldew	336057	534415	222	14	2	31	33	0	2	8	Boulders			0	Steep	None	0	1	0	Improved grassland
09/09/2004	Caldew	336309	535228	211	15	0	24	24	0	0	9	Boulders			0	Steep	None	1	1	0	Improved grassland
09/09/2004	Caldew	336729	536501	194	14	0	26	26	0	11	8	Cobbles	Boulders		0	Shallow	None	0	1	1	Improved grassland
27/09/2004	Caldew	335094	540244	184	12	0	1	1	0	2	0	Cobbles	Sand		0	Steep	None	0	0	0	Woodland
25/08/2004	Caldew (Grainsgill)	332519	532848	328	15	1	0	1	5	0	4	Boulders	Bedrock	Cobbles	0	Steep	None	1	0	0	Unimproved grassland
26/08/2004	Cambeck	351876	565289	328	12	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	0	Steep	Earth cliff	0	1	5	Improved grassland
26/08/2004	Cambeck	352629	567560	0	12	2	0	2	0	0	0	Cobbles	Pebbles	Gravel	0	Steep	None	0	1	0	Improved grassland
26/08/2004	Cambeck	353515	568841	0	12	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	0	Steep	None	0	1	3	Improved grassland
26/08/2004	Cambeck	351141	564332	0	13	0	0	0	0	0	0	Cobbles	Pebbles	Bedrock	0	Steep	None	0	0	0	Woodland

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						No. of fish caught in 5 minutes of fishing															
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26/08/2004	Cambeck	350972	562672	0	13	0	34	34	0	7	5	Pebbles	Cobbles	Gravel	0	Shallow	None	0	0	0	Improved grassland
26/08/2004	Cambeck	351021	563503	0	14	6	1	7	1	2	1	Cobbles	Pebbles	Gravel	0	Steep	None	0	0	0	Woodland
08/10/2004	Castle beck	355829	562886	69	11	10	0	10	6	2	2	Pebbles	Cobbles		0	Steep	None	0	0	0	Woodland
08/10/2004	Castle beck	357051	562333	134	11	1	0	1	9	0	0	Bedrock	Cobbles	Pebbles	0	Steep	None	0	1	0	Woodland
03/09/2004	Croglin	355796	545807	146	14	2	0	2	6	0	1	Cobbles	Pebbles	Gravel	0	Shallow	None	1	0	0	Improved grassland
03/09/2004	Croglin	358648	547517	268	13	0	0	0	9	0	0	Boulders	Bedrock	Cobbles	0	Shallow	None	1	0	0	Unimproved grassland
03/09/2004	Croglin	357562	546991	200	13	1	0	1	6	0	1	Cobbles	Boulders	Pebbles	0	Steep	None	0	1	0	Woodland
03/09/2004	Croglin	356301	546536	181	14	1	0	1	10	0	0	Cobbles	Boulders	Bedrock	0	Steep	Earth cliff	1	0	0	Improved grassland
01/10/2004	Croglin	354671	544410	44	10	12	0	12	4	0	5	Cobbles	Pebbles		0	Shallow	Poached	1	0	0	Improved grassland
01/10/2004	Croglin	353868	542786	54	11	3	0	3	0	0	1	Pebbles	Cobbles	Gravel	0	Steep	None	0	1	4	Improved grassland
01/09/2004	Crowdundle	360842	528158	134	13	0	36	36	0	3	16	Cobbles	Pebbles	Gravel	0	Steep	None	0	0	10	Improved grassland
01/09/2004	Crowdundle	365634	528530	180	10	21	20	41	1	6	17	Cobbles	Pebbles	Gravel	0	Shallow	Earth cliff	0	0	0	Improved grassland
01/09/2004	Crowdundle	364494	530545	179	12	9	25	34	5	6	15	Cobbles	Boulders	Pebbles	0	Steep	Poached	0	0	0	Improved grassland
01/09/2004	Crowdundle	363024	529049	130	12	15	33	48	0	4	15	Cobbles	Boulders	Pebbles	0	Shallow	None	0	0	3	Improved grassland
05/09/2004	Dacre	347709	526752	142	12	0	11	11	0	2	5	Cobbles	Pebbles	Gravel	0	Steep	Poached	1	0	0	Improved grassland
08/09/2004	Dacre	346432	526240	154	13	0	25	25	0	7	8	Bedrock	Cobbles	Pebbles	0	Steep	Undercut	1	0	0	Improved grassland
08/09/2004	Dacre	345331	526103	155	13	0	6	6	2	1	3	Bedrock	Cobbles	Pebbles	0	Steep	None	0	1	0	Woodland
01/10/2004	Dacre (Skitwith)	343402	526667	216	10	1	30	31	0	14	8	Cobbles	Boulders	Pebbles	0	Steep	None	0	0	2	Park/garden
01/10/2004	Dacre (Skitwith)	342137	526917	261	11	3	9	12	7	8	4	Boulders	Cobbles		0	Shallow	None	0	0	1	Improved grassland
01/10/2004	Dacre (Thackthwaite)	343401	526528	212	10	2	55	57	1	8	8	Cobbles	Pebbles		0	Steep	None	0	1	2	Improved grassland
01/10/2004	Dacre (Thackthwaite)	342176	525986	242	11	3	59	62	0	0	9	Pebbles	Cobbles	Gravel	1	Shallow	Earth cliff	0	0	5	Improved grassland
01/10/2004	Dacre (Thackthwaite)	341104	525040	261	11	11	13	24	3	3	5	Gravel	Sand	Silt/mud	0	Steep	None	1	0	2	Improved grassland
24/09/2004	Deepdale	338080	512631	265	8	4	0	4	4	0	1	Bedrock	Boulders		0	Steep	Undercut	0	0	0	Unimproved grassland
24/09/2004	Deepdale	338543	512875	268	9	6	0	6	0	0	2	Boulders	Bedrock	Cobbles	0	Shallow	None	0	0	0	Unimproved grassland
24/09/2004	Deepdale	339089	513355	223	10	3	0	3	1	0	1	Boulders	Cobbles		0	Shallow	Undercut	0	0	0	Unimproved grassland
24/09/2004	Deepdale	339924	514298	162	11	2	1	3	1	1	1	Bedrock	Boulders	Cobbles	0	Steep	Poached	1	0	2	Improved grassland
06/09/2004	Gelt	354290	556125	139	14	8	5	13	1	4	7	Cobbles	Pebbles	Gravel	0	Steep	None	0	1	0	Woodland
06/09/2004	Gelt	353167	557475	139	14	9	19	28	1	7	12	Cobbles	Bedrock	Pebbles	0	Steep	None	0	1	0	Woodland
06/09/2004	Gelt	352703	558572	139	14	2	18	20	0	24	9	Boulders	Bedrock	Cobbles	0	Steep	None	0	0	0	Woodland
29/07/2004	Gelt	357604	551656	180	19	0	0	0	2	12	0	Bedrock	Boulders	Cobbles	0	Steep	Undercut	0	0	0	Unimproved grassland
29/07/2004	Gelt	358430	553183	180	19	2	0	2	11	0	0	Bedrock	Boulders	Pebbles	0	Steep	None	0	0	0	Unimproved grassland
29/07/2004	Gelt	357407	553849	222	15	0	0	0	9	10	0	Boulders	Bedrock	Cobbles	0	Steep	None	0	1	0	Woodland
06/09/2004	Gelt	351309	559397	139	14	2	11	13	0	12	5	Cobbles	Boulders	Pebbles	0	Steep	None	0	1	3	Improved grassland
02/09/2004	Glassonby beck	357637	539416	110	12	16	1	17	2	3	6	Cobbles	Pebbles	Sand	1	Steep	Undercut	0	1	0	Park/garden
02/09/2004	Glassonby beck	358878	539716	109	12	30	0	30	5	0	7	Cobbles	Pebbles	Gravel	0	Steep	None	0	1	0	Woodland
02/09/2004	Glassonby beck	358591	539561	90	13	0	34	34	0	5	6	Cobbles	Pebbles	Gravel	0	Steep	Earth cliff	1	0	0	Improved grassland
16/09/2004	Glencoyne	338648	518602	365	12	5	0	5	2	3	2	Cobbles	Boulders		0	Shallow	Undercut	0	0	5	Improved grassland
26/09/2004	Glencoyne	337879	518633	203	12	32	0	32	11	0	3	Boulders	Cobbles		0	Steep	None	1	0	0	Unimproved grassland
26/09/2004	Glencoyne	338246	518692	184	12	11	0	11	4	1	1	Boulders	Cobbles		0	Steep	None	1	0	0	Unimproved grassland
29/09/2004	Glenridding	336728	517362	250	10	0	0	0	2	0	0	Bedrock	Boulders	Cobbles	0	Shallow	None	0	0	20	Unimproved grassland
29/09/2004	Glenridding	337314	517197	235	11	1	0	1	3	0	0	Boulders	Bedrock	Cobbles	0	Shallow	None	0	0	20	Unimproved grassland
29/09/2004	Glenridding	337763	516968	207	11	4	0	4	4	0	2	Boulders	Bedrock	Cobbles	0	Shallow	None	0	0	20	Improved grassland
29/09/2004	Glenridding	338470	516857	172	11	2	0	2	1	0	1	Boulders	Bedrock	Cobbles	0	Shallow	None	0	0	0	Park/garden

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Date of survey	Sub-catchment	Easting	Northing	Altitude (m above sea level)	Temp °C	No. of fish caught in 5 minutes of fishing: 0+ Trout	0+ Salmon	Total 0+ salmon & trout	1+ and older Trout	1+ and older Salmon	No of 0+ salmon & trout missed	Dominant substrate	Substrate2	Substrate3	Siltation 1=present 0=absent	Bank profile	Erosion type	Stock access 1=yes 0=no	Tunnelling 1=present 0=absent	Fence width (m)	Landuse
29/09/2004	Glenridding	338962	517007	171	12	1	2	3	2	2	1	Cobbles	Pebbles	Gravel	0	Shallow	None	0	0	0	Park/garden
26/09/2004	Goldrill	340254	513410	173	12	6	5	11	1	1	4	Cobbles	Pebbles		0	Steep	None	0	0	0	Woodland
26/09/2004	Goldrill	340428	514474	158	13	21	7	28	1	1	5	Pebbles	Cobbles		0	Steep	None	0	0	4	Unimproved grassland
26/09/2004	Goldrill	339938	515496	160	13	9	7	16	2	0	5	Cobbles	Pebbles		0	Steep	None	0	0	2	Unimproved grassland
02/08/2004	Grisedale	338942	516040	199	19	6	0	6	2	8	2	Cobbles	Boulders	Pebbles	0	Steep	None	0	1	0	Woodland
02/08/2004	Grisedale	339333	516276	160	18	0	18	18	0	0	8	Cobbles	Pebbles	Gravel	0	Steep	None	0	0	1	
02/08/2004	Grisedale	338675	514732	237	19	4	0	4	1	2	2	Cobbles	Boulders	Pebbles	0	Shallow	None	1	0	0	Unimproved grassland
02/08/2004	Grisedale	337881	515482	199	22	0	0	0	2	3	0	Cobbles	Pebbles		0	Shallow	None	0	0	0	Improved grassland
02/08/2004	Grisedale	337386	515157	200	20	2	0	2	5	11	1	Cobbles	Boulders	Pebbles	0	Steep	None	1	0	0	Unimproved grassland
02/08/2004	Grisedale Beck	336238	514345	232	15	0	0	0	2	8	0	Cobbles	Bedrock	Boulders	0	Steep	None	1	0	0	Unimproved grassland
23/08/2004	Hartley	377688	509012	179	12	37	37	74	13	9	17	Cobbles	Pebbles		0	Steep	Undercut	0	0	10	Improved grassland
23/08/2004	Hartley	378293	508747	186	11	63	0	63	14	0	12	Cobbles	Pebbles	Gravel	0	Shallow	None	0	0	0	Park/garden
25/09/2004	Haweswater	351050	515987	250	13	0	0	0	4	0	0	Cobbles	Boulders		0	Shallow	None	1	0	0	Unimproved grassland
25/09/2004	Haweswater	351835	518034	250	13	3	21	24	1	5	2	Pebbles			0	Steep	Undercut	0	1	0	Improved grassland
25/09/2004	Haweswater	351199	516057	250	13	0	7	7	0	19	5	Boulders	Bedrock		0	Shallow	None	1	0	0	Unimproved grassland
24/09/2004	Hayeswater	340744	513107	186	12	9	2	11	0	4	5	Cobbles	Pebbles	Gravel	0	Steep	None	0	0	0	Improved grassland
24/09/2004	Hayeswater	341575	512710	201	12	3	0	3	5	0	1	Pebbles	Boulders	Cobbles	0	Steep	Poached	1	0	0	Unimproved grassland
24/09/2004	Hayeswater (Pasture)	341605	512630	206	12	5	0	5	1	4	2	Boulders	Cobbles	Pebbles	0	Steep	Poached	1	0	0	Unimproved grassland
12/07/2004	Helm	370988	514881	167	12	0	1	1	1	0	0	Pebbles	Gravel	Silt/mud	1	Shallow	Earth cliff	1	0	0	Improved grassland
12/07/2004	Helm	370898	513871	167	13	0	0	0	1	2	0	Cobbles	Pebbles		1	Steep	None	0	0	2	Improved grassland
12/07/2004	Helm	371309	512161	192	12	7	0	7	1	0	1	Gravel	Pebbles	Sand	1	Shallow	Poached	0	0	0	Improved grassland
12/07/2004	Helm	370512	508507	246	16	2	0	2	1	0	0	Gravel	Pebbles		0	Shallow	None	1	0	2	Unimproved grassland
12/07/2004	Helm	370293	516696	160	14	0	12	12	0	1	0	Cobbles	Pebbles	Gravel	0	Steep	None	0	1	2	Improved grassland
12/07/2004	Helm	371359	510995	196	13	21	0	21	0	0	13	Pebbles	Gravel	Sand	0	Steep	None	0	0	1	Improved grassland
12/07/2004	Helm	370886	510064	194	14	2	0	2	0	0	0	Pebbles	Cobbles		1	Shallow	Poached	1	1	0	Improved grassland
12/07/2004	Helm	370644	509304	230	16	3	0	3	1	0	0	Cobbles	Boulders	Pebbles	0	Shallow	None	1	0	0	Unimproved grassland
13/07/2004	Helm	371058	516881	140	13	0	6	6	0	8	0	Pebbles	Gravel		0	Shallow	Earth cliff	1	0	0	Improved grassland
28/09/2004	Heltondale	350228	520877	242	13	23	0	23	3	7	10	Boulders	Cobbles	Pebbles	0	Steep	None	0	1	0	Woodland
28/09/2004	Heltondale	350987	520286	188	14	4	43	47	0	6	19	Cobbles	Boulders	Gravel	0	Steep	None	0	0	0	Improved grassland
28/09/2004	Heltondale	350688	520574	192	14	4	8	12	3	10	7	Pebbles	Cobbles	Boulders	0	Steep	None	1	0	0	Unimproved grassland
13/07/2004	Hilton	375596	521874	313	13	35	1	36	8	54	3	Cobbles	Boulders	Pebbles	0	Shallow	None	1	0	0	Unimproved grassland
13/07/2004	Hilton	375194	521362	300	15	4	0	4	12	30	0	Boulders	Pebbles	Sand	0	Shallow	Earth cliff	1	0	0	Unimproved grassland
13/07/2004	Hilton	374139	521079	255	16	5	0	5	5	34	0	Boulders	Cobbles	Pebbles	0	Shallow	Undercut	0	0	0	Unimproved grassland
14/07/2004	Hilton	372449	520439	255	13	16	6	22	5	34	3	Cobbles	Boulders	Pebbles	0	Steep	Earth cliff	0	0	1	Unimproved grassland
14/07/2004	Hilton	371273	529976	185	14	3	19	22	1	12	3	Cobbles	Pebbles	Gravel	0	Shallow	Earth cliff	1	0	0	Improved grassland
14/07/2004	Hilton (Coupland)	370755	518806	152	15	5	17	22	2	3	4	Gravel	Pebbles	Gravel	0	Steep	None	0	1	1	Improved grassland
13/07/2004	Hilton Swindale	375371	521426	304	15	6	0	6	9	0	1	Boulders	Cobbles	Pebbles	0	Steep	Earth cliff	1	0	0	Unimproved grassland
15/07/2004	Hoff	368142	515356	172	14	1	0	1	0	0	0	Cobbles	Pebbles	Gravel	1	Steep	Earth cliff	1	0	1	Improved grassland
15/07/2004	Hoff	368254	516000	164	14	3	22	25	1	6	4	Bedrock	Pebbles	Gravel	1	Shallow	Poached	1	0	0	Improved grassland
15/07/2004	Hoff	361179	519008	151	16	1	15	16	0	2	0	Bedrock	Pebbles		0	Steep	None	0	0	3	Improved grassland
15/07/2004	Hoff	368804	519924	151	16	0	13	13	0	2	5	Cobbles	Boulders	Gravel	0	Steep	None	0	0	5	Improved grassland
14/07/2004	Hoff	367310	514585	192	15	1	0	1	0	0	0	Pebbles	Bedrock	Gravel	0	Steep	Undercut	1	0	0	Improved grassland
16/07/2004	Hoff	368466	520977	144	15	0	17	17	0	2	4	Cobbles	Pebbles	Gravel	0	Steep	None	0	0	0	Improved grassland
15/07/2004	Hoff	367602	517588	148	15	1	6	7	1	6	4	Pebbles	Gravel		0	Shallow	Earth cliff	1	0	0	Improved grassland

2004 Survey: Semi-quantitative electrofishing with habitat data (Eden Rivers Trust)

						No. of fish caught in 5 minutes of fishing															
Date of survey	Sub-catchment	Easting	Northing	Altitude (m above sea level)	Temp °C	0+ Trout	0+ Salmon	Total 0+ salmon & trout	1+ and older Trout	1+ and older Salmon	No of 0+ salmon & trout missed	Dominant substrate	Substrate2	Substrate3	Siltation 1=present 0=absent	Bank profile	Erosion type	Stock access 1=yes 0=no	Tunnelling 1=present 0=absent	Fence width (m)	Landuse
14/07/2004	Hoff (Scale Beck)	365941	513796	155	13	1	0	1	0	0	0	Silt/mud	Pebbles	Gravel	1	Steep	Poached	1	1	0	Unimproved grassland
14/07/2004	Hoff (Scale Beck)	366484	514083	186	18	2	0	2	0	0	0	Pebbles	Gravel	Silt/mud	1	Steep	Undercut	1	0	2	Improved grassland
28/09/2004	Howe	350544	517734	277	11	0	0	0	5	0	0	Cobbles	Boulders	Pebbles	0	Steep	None	0	1	2	Unimproved grassland
28/09/2004	Howe	351676	518125	192	13	5	7	12	8	3	6	Cobbles	Pebbles	Gravel	0	Steep	None	0	1	0	Improved grassland
31/08/2004	Howesgrain	343351	518335	207	14	6	6	12	1	0	6	Pebbles	Cobbles	Gravel	1	Shallow	Poached	1	0	0	Unimproved grassland
18/08/2004	Irthing	364094	577644	282	15	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	0	Shallow	Undercut	0	0	0	Unimproved grassland
18/08/2004	Irthing	368054	577487	279	14	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	0	Shallow	None	0	0	3	Unimproved grassland
11/10/2004	Irthing	368013	574323	244	8	0	0	0	2	0	0	Cobbles	Pebbles		0	Shallow	None	1	0	5	Unimproved grassland
11/10/2004	Irthing	366442	571083	239	9	0	0	0	0	0	0	Cobbles	Pebbles		0	Shallow	None	1	0	10	Unimproved grassland
11/10/2004	Irthing	363831	567684	180	9	0	2	2	0	4	0	Bedrock	Boulders	Cobbles	0	Steep	None	0	0	0	Woodland
11/10/2004	Irthing	362225	566552	107	10	0	29	29	2	3	8	Boulders	Cobbles	Pebbles	0	Steep	Earth cliff	1	0	0	Improved grassland
11/10/2004	Irthing	359691	564777	91	10	0	19	19	1	3	7	Cobbles	Boulders		0	Steep	None	1	0	0	Improved grassland
11/10/2004	Irthing	358567	564398	80	10	0	10	10	9	0	3	Boulders	Cobbles		0	Steep	None	0	0	10	Improved grassland
12/10/2004	Irthing	350353	560998	30	10	0	10	10	0	0	5	Bedrock	Cobbles	Pebbles	1	Shallow	None	0	0	0	Arable crop
12/10/2004	Irthing	349253	560254	25	10	0	22	22	0	0	3	Pebbles	Cobbles	Gravel	1	Steep	None	0	0	0	Woodland
12/10/2004	Irthing	349138	558520	29	10	0	11	11	0	0	2	Cobbles	Pebbles		1	Steep	None	0	0	5	Improved grassland
12/10/2004	Irthing	358457	563944	78	9	0	10	10	0	5	3	Boulders	Cobbles		0	Steep	None	0	1	0	Woodland
12/10/2004	Irthing	354532	563199	78	9	0	10	10	0	4	5	Boulders	Cobbles	Pebbles	1	Steep	None	0	0	0	Improved grassland
12/10/2004	Irthing	351220	562323	30	10	0	19	19	7	0	8	Cobbles	Pebbles	Gravel	0	Steep	None	0	0	6	Improved grassland
18/08/2004	Irthing (Butterburn)	367150	574621	251	16	2	0	2	0	0	0	Cobbles	Boulders	Pebbles	0	Shallow	None	1	0	0	Unimproved grassland
11/10/2004	Irthing (Butterburn)	367878	574144	241	8	0	0	0	2	0	0	Cobbles			0	Shallow	Undercut	1	0	0	Unimproved grassland
23/07/2004	Kingwater	353182	564907	66	16	0	3	3	1	2	2	Bedrock	Cobbles	Pebbles	0	Steep	Earth cliff	0	0	5	Improved grassland
23/07/2004	Kingwater	353918	565762	72	16	1	2	3	0	0	2	Cobbles	Pebbles	Gravel	1	Steep	Earth cliff	1	0	0	Improved grassland
23/07/2004	Kingwater	356428	567246	93	16	0	1	1	0	1	0	Cobbles	Boulders	Pebbles	1	Steep	Earth cliff	1	1	0	Improved grassland
23/07/2004	Kingwater	357838	567495	91	16	0	4	4	0	0	0	Cobbles	Boulders	Pebbles	0	Steep	None	0	0	5	Improved grassland
29/07/2004	Kingwater	362898	572089	253	17	0	0	0	1	0	0	Cobbles	Pebbles	Silt/mud	0	Steep	None	0	0	0	Conifer plantation
29/07/2004	Kingwater	361702	569967	185	18	0	0	0	3	0	0	Bedrock	Boulders	Cobbles	0	Steep	None	0	0	0	Conifer plantation
29/07/2004	Kingwater	359491	568653	185	18	0	0	0	3	0	0	Bedrock	Boulders	Cobbles	0	Steep	None	0	1	0	Woodland
23/07/2004	Kingwater	355307	566708	70	16	3	22	25	1	3	7	Cobbles	Pebbles	Gravel	0	Steep	Earth cliff	0	0	0	Improved grassland
17/08/2004	Kirkland (Briggle)	365298	532730	239	14	16	0	16	4	0	6	Bedrock	Boulders	Cobbles	0	Steep	None	1	0	0	Unimproved grassland
17/08/2004	Kirkland (Briggle)	364469	532475	239	15	9	0	9	11	0	3	Cobbles	Pebbles	Sand	1	Shallow	Poached	1	0	0	Unimproved grassland
26/09/2004	Kirkstone	339937	511496	185	11	10	0	10	1	3	3	Cobbles	Pebbles		0	Shallow	None	1	0	1	Unimproved grassland
26/09/2004	Kirkstone	339962	511880	175	12	11	0	11	1	7	3	Pebbles	Cobbles		0	Steep	None	0	0	2	Improved grassland
26/09/2004	Kirkstone (Dovedale)	339741	511723	203	12	8	0	8	0	1	4	Cobbles	Pebbles		0	Shallow	Poached	1	0	0	Unimproved grassland
24/08/2004	Ladthwaite	380000	506733	342	12	0	0	0	21	0	0	Boulders	Cobbles	Pebbles	0	Steep	Undercut	0	0	0	Unimproved grassland
24/08/2004	Ladthwaite	379571	506875	342	13	5	0	5	4	0	2	Cobbles	Pebbles	Gravel	0	Steep	Undercut	0	0	2	Unimproved grassland
03/08/2004	Leith	358606	525211	135	18	0	29	29	0	2	6	Cobbles	Pebbles	Boulders	1	Steep	Poached	1	0	0	Improved grassland
03/08/2004	Leith	355749	525047	133	17	0	5	5	0	0	2	Cobbles	Pebbles	Gravel	0	Shallow	None	0	1	0	Woodland
03/08/2004	Leith	355509	524096	133	16	1	2	3	0	0	1	Cobbles	Boulders	Bedrock	0	Shallow	None	0	1	0	Unimproved grassland
03/08/2004	Leith	355118	524061	133	18	1	0	1	0	3	0	Cobbles	Boulders	Bedrock	0	Shallow	None	0	0	0	Improved grassland

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Date of survey	Sub-catchment	Easting	Northing	Altitude (m above sea level)	Temp °C	No. of fish caught in 5 minutes of fishing: 0+ Trout	0+ Salmon	Total 0+ salmon & trout	1+ and older Trout	1+ and older Salmon	No of 0+ salmon & trout missed	Dominant substrate	Substrate2	Substrate3	Siltation 1=present 0=absent	Bank profile	Erosion type	Stock access 1=yes 0=no	Tunnelling 1=present 0=absent	Fence width (m)	Landuse
03/08/2004	Leith	355103	522605	182	17	2	0	2	2	4	0	Cobbles	Pebbles		0	Shallow	None	0	1	0	Improved grassland
03/08/2004	Leith	355247	521657	198	16	4	0	4	5	1	0	Cobbles	Boulders		0	Shallow	None	0	1	0	Woodland
04/08/2004	Leith	355811	520679	220	15	5	0	5	0	0	9	Cobbles	Boulders		0	Shallow	None	1	0	0	Improved grassland
05/08/2004	Leith	355949	518755	233	16	13	0	13	8	0	6	Pebbles	Cobbles		1	Shallow	Poached	1	0	0	Improved grassland
05/08/2004	Leith	355563	518287	235	16	4	0	4	6	0	1	Pebbles	Cobbles	Gravel	1	Shallow	None	0	1	0	Woodland
05/08/2004	Leith	355937	519462	235	15	13	0	13	0	0	2	Pebbles	Cobbles	Gravel	0	Steep	Poached	1	1	0	Improved grassland
05/08/2004	Leith	355752	520369	211	17	3	0	3	0	0	1	Cobbles	Pebbles		0	Shallow	Poached	1	0	0	Improved grassland
30/07/2004	Leith	358706	524415	118	19	0	0	0	0	0	0	Cobbles	Pebbles	Silt/mud	0	Steep	Poached	1	0	0	Improved grassland
30/07/2004	Leith	357587	524755	119	20	4	3	7	1	0	2	Cobbles	Pebbles	Gravel	0	Steep	None	0	0	0	
05/08/2004	Leith	355776	519910	230	14	3	0	3	1	0	1	Cobbles	Boulders		0	Shallow	Poached	1	0	0	Unimproved grassland
25/09/2004	Lowther	355990	512631	244	11	3	17	20	1	10	5	Cobbles	Boulders		0	Steep	None	0	0	0	Unimproved grassland
13/10/2004	Lowther	352619	528646	121	10	0	14	14	8	0	6	Cobbles	Pebbles	Gravel	0	Shallow	None	0	0	5	Improved grassland
13/10/2004	Lowther	352000	525024	123	10	0	11	11	1	5	5	Cobbles	Boulders	Pebbles	0	Shallow	None	1	0	0	Improved grassland
13/10/2004	Lowther	351858	522177	181	10	1	9	10	0	0	4	Cobbles	Pebbles		0	Shallow	None	1	0	0	Improved grassland
13/10/2004	Lowther	351493	519704	180	10	0	5	5	1	2	2	Pebbles	Gravel		0	Shallow	Earth cliff	1	0	0	Unimproved grassland
13/10/2004	Lowther	353439	516451	194	10	0	7	7	0	6	5	Boulders	Cobbles		0	Shallow	None	1	0	0	Improved grassland
13/10/2004	Lowther	355131	514333	218	10	0	32	32	1	2	6	Cobbles	Pebbles	Gravel	0	Shallow	None	1	0	0	Unimproved grassland
25/09/2004	Lowther (Swindale)	351479	513221	250	12	1	20	21	1	4	6	Cobbles			0	Shallow	None	1	0	0	Unimproved grassland
25/09/2004	Lowther (Swindale)	352695	515029	221	11	0	75	75	0	17	15	Cobbles	Boulders		0	Shallow	None	1	0	0	Unimproved grassland
25/09/2004	Lowther (Swindale)	350386	511972	235	11	0	36	36	0	5	8	Cobbles	Bedrock	Boulders	0	Shallow	Earth cliff	1	0	0	Improved grassland
28/09/2004	Lowther (Wet Sleddale)	354036	511126	187	14	1	0	1	3	0	0	Bedrock	Boulders	Cobbles	0	Steep	None	0	0	5	Unimproved grassland
27/07/2004	Lyvennet	362371	514854	170	15	6	0	6	2	0	3	Cobbles	Pebbles	Boulders	0	Steep	None	0	0	0	Improved grassland
27/07/2004	Lyvennet	362619	516050	175	16	18	0	18	1	0	6	Bedrock	Cobbles	Pebbles	0	Shallow	None	1	0	0	Improved grassland
26/07/2004	Lyvennet	361614	511419	292	13	12	0	12	0	0	5	Bedrock	Cobbles	Pebbles	1	Shallow	Poached	1	0	0	Unimproved grassland
26/07/2004	Lyvennet	362102	512679	213	14	1	0	1	3	0	0	Cobbles	Boulders		0	Shallow	Poached	1	0	0	Improved grassland
26/07/2004	Lyvennet	362278	513309	209	14	1	0	1	2	0	0	Cobbles	Bedrock	Boulders	1	Shallow	Poached	1	1	0	Improved grassland
27/07/2004	Lyvennet	362476	516896	175	18	0	9	9	0	1	5	Cobbles	Pebbles		1	Shallow	Poached	1	0	0	Improved grassland
27/07/2004	Lyvennet	362111	517756	175	14	2	7	9	1	2	3	Cobbles	Boulders	Pebbles	0	Steep	None	0	1	100	Conifer plantation
27/07/2004	Lyvennet	362134	518880	175	14	1	20	21	0	0	8	Pebbles	Cobbles	Gravel	0	Steep	None	0	0	0	Improved grassland
27/07/2004	Lyvennet	361940	520082	171	18	0	6	6	0	0	3	Pebbles	Cobbles	Gravel	0	Steep	None	0	0	5	Improved grassland
28/07/2004	Lyvennet	361759	520734	171	16	1	2	3	0	1	1	Bedrock	Cobbles	Pebbles	0	Shallow	Poached	0	0	0	Improved grassland
28/07/2004	Lyvennet	361458	521943	168	17	0	0	0	0	2	0	Cobbles	Bedrock	Pebbles	0	Steep	None	0	0	2	Improved grassland
28/07/2004	Lyvennet	361224	523134	168	17	0	4	4	0	1	1	Bedrock	Boulders	Cobbles	0	Steep	None	0	1	0	Arable crop
28/07/2004	Lyvennet	360892	523845	161	20	0	2	2	0	2	0	Cobbles	Boulders	Pebbles	0	Shallow	Poached	1	0	4	Improved grassland
28/07/2004	Lyvennet	360584	524071	135	20	0	12	12	0	1	4	Bedrock	Boulders	Cobbles	0	Steep	None	0	0	0	Improved grassland
28/07/2004	Lyvennet	360029	524680	121	20	0	8	8	1	2	3	Pebbles	Cobbles	Gravel	0	Steep	None	0	0	5	Improved grassland
28/07/2004	Lyvennet	360525	525091	118	20	0	26	26	0	0	5	Pebbles	Cobbles		0	Shallow	Poached	1	0	0	Improved grassland
30/07/2004	Lyvennet	361173	524952	111	17	0	1	1	0	12	0	Cobbles	Pebbles	Gravel	0	Shallow	None	0	0	10	Arable crop
30/07/2004	Lyvennet	360839	525907	110	18	0	21	21	0	0	6	Pebbles	Pebbles	Cobbles	0	Shallow	None	0	0	10	Improved grassland
26/07/2004	Lyvennet	362302	513885	208	16	0	0	0	1	0	0	Cobbles	Boulders	Pebbles	0	Steep	None	1	0	0	Improved grassland
01/09/2004	Milburn (Crowdundle)	364632	528918	177	10	5	10	15	4	4	6	Cobbles	Pebbles	Gravel	0	Shallow	Earth cliff	0	0	0	Improved grassland

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08/10/2004	Milton/Quarry beck	356797	560227	150	10	1	0	1	7	0	0	Pebbles	Cobbles	Silt/mud	1	Shallow	None	1	0	0	Improved grassland
08/10/2004	Milton/Quarry beck	355304	560622	150	10	6	0	6	16	0	2	Pebbles	Cobbles	Sand	1	Steep	Poached	1	0	0	Improved grassland
08/10/2004	Milton/Quarry beck	354772	561083	123	10	1	0	1	13	0	0	Pebbles	Cobbles	Sand	1	Shallow	Earth cliff	1	0	0	Improved grassland
28/07/2004	Morland Beck	360615	523804	122	18	0	5	5	0	0	2	Cobbles	Boulders	Pebbles	0	Steep	None	0	0	0	Improved grassland
16/08/2004	Pettril	345362	532722	184	12	7	0	7	1	0	2	Cobbles	Pebbles	Gravel	0	Shallow	None	0	0	5	Woodland
16/08/2004	Pettril	344996	531970	184	12	9	0	9	4	0	4	Cobbles	Pebbles		0	Steep	Undercut	0	0	3	Arable crop
16/08/2004	Pettril	344688	530542	193	13	5	0	5	3	0	2	Pebbles	Gravel	Sand	1	Steep	Undercut	0	0	2	Improved grassland
16/08/2004	Pettril	346834	532727	192	14	1	0	1	0	0	0	Pebbles	Cobbles	Boulders	0	Steep	None	0	1	3	Improved grassland
16/08/2004	Pettril	348965	532903	155	14	0	0	0	0	0	0	Cobbles	Pebbles		1	Shallow	Poached	1	0	0	Improved grassland
16/08/2004	Pettril	347872	531907	197	14	2	0	2	1	0	0	Cobbles	Pebbles	Gravel	0	Shallow	Undercut	0	0	0	Park/garden
30/09/2004	Pettril	349589	533532	170	11	0	0	0	1	0	0	Cobbles	Pebbles	Gravel	0	Steep	Poached	1	0	0	Improved grassland
30/09/2004	Pettril	349285	535793	132	12	0	0	0	1	0	0	Pebbles	Gravel	Sand	1	Steep	Earth cliff	1	0	0	Improved grassland
30/09/2004	Pettril	348469	539665	118	12	0	0	0	1	0	0	Bedrock	Boulders	Cobbles	1	Steep	Poached	1	0	0	Improved grassland
30/09/2004	Pettril	349102	538767	130	12	0	0	0	0	0	0	Pebbles	Gravel	Sand	1	Shallow	None	0	0	20	Improved grassland
30/09/2004	Pettril	347478	540738	116	13	0	0	0	0	0	0	Pebbles	Gravel	Sand	1	Shallow	Poached	1	0	0	Improved grassland
30/09/2004	Pettril	348360	542742	102	12	0	0	0	0	0	0	Pebbles	Gravel	Sand	1	Steep	None	0	0	20	Improved grassland
30/09/2004	Pettril	345062	545106	93	13	0	0	0	0	0	0	Pebbles	Cobbles	Gravel	1	Shallow	Poached	1	0	5	Improved grassland
30/09/2004	Pettril	344195	548097	66	13	0	0	0	0	0	0	Pebbles	Gravel	Sand	1	Steep	None	0	0	0	Woodland
30/09/2004	Pettril	343521	551269	44	13	0	0	0	0	3	0	Cobbles	Pebbles	Gravel	1	Steep	None	0	0	15	Improved grassland
08/10/2004	Quarry beck	354851	562361	123	10	0	0	0	0	0	0	Bedrock	Pebbles	Sand	1	Steep	Undercut	1	0	0	Woodland
08/10/2004	Quarry beck	355317	563249	73	10	1	21	22	3	4	4	Bedrock	Pebbles	Gravel	0	Steep	None	0	0	0	Woodland
31/08/2004	Rampsgill	343821	516362	221	13	33	8	41	2	0	14	Pebbles	Cobbles	Gravel	0	Shallow	None	1	0	0	Unimproved grassland
02/09/2004	Ravenbeck	358595	541886	87	13	4	0	4	5	0	2	Cobbles	Pebbles	Gravel	0	Steep	Undercut	0	0	5	Improved grassland
02/09/2004	Ravenbeck	356668	541610	80	13	1	0	1	6	0	0	Cobbles	Boulders	Pebbles	0	Steep	Poached	1	0	0	Improved grassland
03/09/2004	Ravenbeck	355277	541076	86	12	1	46	47	0	3	16	Cobbles	Boulders	Pebbles	0	Steep	None	0	0	0	Park/garden
03/09/2004	Ravenbeck	359202	542592	162	12	17	0	17	18	0	5	Cobbles	Boulders	Pebbles	0	Shallow	None	0	0	0	Woodland
03/09/2004	Ravenbeck	360175	543049	162	12	14	0	14	15	0	3	Boulders	Bedrock	Cobbles	0	Steep	None	0	1	0	Woodland
02/09/2004	Robbery Water	358586	535968	89	12	3	25	28	2	4	11	Cobbles	Boulders	Pebbles	1	Steep	Poached	1	0	0	Improved grassland
02/09/2004	Robbery Water	357952	535411	80	12	12	0	12	2	1	6	Cobbles	Pebbles	Sand	1	Steep	Poached	1	1	0	Improved grassland
31/08/2004	Sandwick	342486	519491	155	14	6	7	13	0	1	5	Boulders	Cobbles	Pebbles	0	Steep	Poached	1	0	0	Improved grassland
19/07/2004	Scandal	376510	511237	160	13	0	37	37	0	0	5	Pebbles	Gravel	Sand	1	Shallow	Poached	1	0	0	Improved grassland
19/07/2004	Scandal	375396	511035	173	15	0	29	29	0	0	0	Bedrock	Pebbles	Gravel	0	Steep	None	1	0	0	Improved grassland
19/07/2004	Scandal	374665	510416	168	14	4	18	22	1	3	7	Cobbles	Pebbles	Gravel	0	Steep	None	0	0	0	Improved grassland
20/07/2004	Scandal	372077	505953	220	20	1	1	2	0	3	0	Cobbles	Boulders	Pebbles	1	Steep	None	0	1	4	Improved grassland
20/07/2004	Scandal	371882	504803	220	16	5	0	5	1	0	2	Cobbles	Boulders	Pebbles	1	Steep	None	0	1	4	Improved grassland
20/07/2004	Scandal	373453	508395	191	14	5	8	13	0	6	5	Bedrock	Pebbles	Gravel	1	Steep	None	1	1	0	Improved grassland
20/07/2004	Scandal	372852	507403	210	15	4	8	12	0	1	4	Cobbles	Pebbles	Gravel	0	Steep	Poached	1	1	0	Improved grassland
20/07/2004	Scandal	372509	506562	220	18	0	3	3	0	2	0	Cobbles	Boulders	Pebbles	0	Steep	None	0	0	2	Improved grassland
20/07/2004	Scandal	372047	504287	247	14	1	0	1	4	0	0	Bedrock	Boulders	Cobbles	1	Steep	None	1	0	0	Improved grassland
19/07/2004	Scandal	373924	509106	187	16	2	31	33	2	0	13	Cobbles	Pebbles	Sand	0	Steep	None	0	0	3	Improved grassland
16/07/2004	Swindale	378484	513980	163	17	0	11	11	2	7	0	Cobbles	Boulders	Pebbles	0	Steep	Earth cliff	1	0	0	Improved grassland
16/07/2004	Swindale	380344	515669	162	16	1	0	1	14	0	0	Boulders	Cobbles		0	Steep	None	0	1	0	Woodland
16/07/2004	Swindale	380695	516070	162	12	1	0	1	8	0	1	Boulders	Cobbles		0	Steep	None	0	1	0	Woodland

2004 Survey: Semi-quantitative electrofishing with habitat data (Eden Rivers Trust)

Date of survey	Sub-catchment	Easting	Northing	Altitude (m above sea level)	Temp °C	No. of fish caught in 6 minutes of fishing: 0+ Trout	0+ Salmon	Total 0+ salmon & trout	1+ and older Trout	1+ and older Salmon	No of 0+ salmon & trout missed	Dominant substrate	Substrate2	Substrate3	Siltation 1=present 0=absent	Bank profile	Erosion type	Stock access 1=yes 0=no	Tunnelling 1=present 0=absent	Fence width (m)	Landuse
16/07/2004	Swindale	377440	513466	0	14	0	13	13	0	2	5	Cobbles	Pebbles	Gravel	0	Steep	Earth cliff	1	0	2	Improved grassland
06/08/2004	Trout	363509	523236	113	16	0	11	11	0	1	6	Pebbles	Cobbles	Gravel	0	Steep	Poached	1	0	0	Improved grassland
06/08/2004	Trout	368406	526987	187	16	43	27	70	1	4	5	Cobbles	Boulders	Pebbles	0	Steep	Undercut	1	1	0	Improved grassland
06/08/2004	Trout	368516	523493	168	17	11	24	35	1	0	8	Pebbles	Cobbles	Gravel	0	Steep	Undercut	1	0	10	Improved grassland
06/08/2004	Trout	368384	526116	187	17	14	52	66	1	3	6	Cobbles	Boulders	Pebbles	0	Steep	Undercut	0	1	0	Improved grassland
06/08/2004	Trout	368574	525426	170	18	13	67	80	0	2	12	Cobbles	Pebbles	Gravel	0	Shallow	Poached	1	0	0	Improved grassland
01/09/2004	Trout	365820	524066	126	13	0	15	15	0	0	8	Cobbles	Boulders	Pebbles	0	Steep	Earth cliff	0	0	10	Improved grassland
06/08/2004	Trout-knock gill	368768	527346	191	17	18	1	19	3	6	5	Cobbles	Boulders	Pebbles	0	Steep	None	0	1	0	Improved grassland
06/08/2004	Trout-knock gill	368921	528105	191	18	3	0	3	3	0	2	Cobbles	Boulders	Pebbles	0	Shallow	None	1	0	0	Unimproved grassland
21/07/2004	Upper Eden	377203	505714	246	18	1	0	1	0	0	0	Cobbles	Pebbles	Gravel	0	Steep	None	0	0	4	Improved grassland
19/07/2004	Upper Eden	376203	513159	148	18	0	2	2	0	0	2	Cobbles	Pebbles	Gravel	0	Shallow	Poached	1	0	0	Improved grassland
21/07/2004	Upper Eden	377816	496542	250	14	1	0	1	1	0	0	Bedrock	Boulders	Cobbles	0	Steep	None	1	1	3	Unimproved grassland
21/07/2004	Upper Eden	377758	498243	250	14	0	0	0	1	0	0	Cobbles	Boulders	Pebbles	0	Steep	None	1	1	0	Improved grassland
21/07/2004	Upper Eden	378178	499434	300	15	1	0	1	0	0	0	Cobbles	Pebbles		0	Shallow	Poached	1	0	0	Improved grassland
21/07/2004	Upper Eden	376002	501309	264	17	0	0	0	1	0	0	Cobbles	Pebbles		0	Steep	None	0	1	0	Improved grassland
21/07/2004	Upper Eden	378120	502830	256	18	1	0	1	0	0	0	Cobbles	Pebbles		0	Steep	None	1	0	1	Improved grassland
17/07/2004	Upper Eden	377630	504486	255	19	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	0	Shallow	Poached	1	0	0	Improved grassland
22/07/2004	Upper Eden	377722	507987	246	16	0	48	48	0	2	8	Cobbles	Gravel	Pebbles	0	Steep	Poached	1	0	0	Improved grassland
22/07/2004	Upper Eden	377434	509320	174	16	2	25	27	0	0	10	Bedrock	Cobbles	Pebbles	0	Steep	Poached	1	0	0	Improved grassland
22/07/2004	Upper Eden	377092	510437	167	16	1	62	63	0	2	10	Cobbles	Pebbles		0	Steep	None	0	0	5	Improved grassland
22/07/2004	Upper Eden	376574	511432	158	17	0	23	23	0	0	15	Pebbles	Cobbles	Gravel	0	Shallow	Poached	1	0	4	Improved grassland
22/07/2004	Upper Eden	377031	512315	153	17	0	12	12	0	0	0	Bedrock	Cobbles	Pebbles	0	Steep	None	0	0	4	Improved grassland
22/07/2004	Upper Eden	375168	513934	143	17	0	25	25	0	0	6	Pebbles	Cobbles		0	Steep	Earth cliff	0	0	0	Improved grassland
22/07/2004	Upper Eden	373539	515951	142	17	0	34	34	0	0	5	Pebbles	Cobbles	Gravel	0	Steep	Earth cliff	1	0	3	Improved grassland
30/07/2004	Upper Eden	361220	525885	98	20	0	6	6	0	0	3	Pebbles	Cobbles	Gravel	0	Shallow	None	1	0	0	Improved grassland

2005 Survey: Semi-quantitative electrofishing with habitat data (Eden Rivers Trust)

Date of survey	Site	Easting	Northing	0+ trout	0+ salmon	Total 0+ salmon & trout	0+ salmon and trout missed	1+ and older trout	1+ and older salmon	Dominant substrate	Substrate 2	Substrate 3	Siltation 1=present 0=absent	Erosion type	Stock access 1=Yes 0=No	Tunnelling 1=present 0=absent	Fence width (m)	Landuse
15-Aug-05	Aira beck	340099	520029	0	1	1	0	1	3	Boulders	Cobbles	Gravel	0	None	0	0	0	Woodland
15-Aug-05	Aira beck	340217	519928	0	10	10	3	0	0	Boulders	Cobbles	Pebbles	0	None	1	0	5	grassland
04-Aug-05	Argill	382537	512910	0	0	0	0	1	0	Bedrock	Boulders	Cobbles	0	None	0	0	0	grassland
15-Aug-05	Bannerdale	343560	516825	12	19	31	15	6	3	Cobbles	Pebbles		1	None	0	0	5	grassland
02-Aug-05	Belah	379232	512040	1	14	15	6	0	8	Boulders	Cobbles	Pebbles	0	Poached	1	0	0	grassland
02-Aug-05	Belah	380254	512345	3	28	31	6	0	7	Boulders	Cobbles	Pebbles	0	None	0	0	5	grassland
02-Aug-05	Belah	377182	512464	0	106	106	20	0	0	Cobbles	Pebbles	Gravel	0	Earth cliff	1	0	0	grassland
04-Aug-05	Belah	382337	511996	6	3	9	3	1	9	Boulders	Cobbles	Gravel	0	None	0	0	0	grassland
02-Aug-05	Belah	378325	511982	0	25	25	6	0	2	Boulders	Cobbles	Pebbles	0	Earth cliff	0	0	5	grassland
15-Aug-05	Boredale	343433	518230	8	0	8	5	2	0	Pebbles	Gravel		1	Poached	1	0	0	grassland
15-Aug-05	Boredale	342624	519137	34	1	35	10	1	0	Cobbles	Boulders	Pebbles	0	None	0	1	5	grassland
03-Aug-05	Briggle	356738	535169	0	39	39	6	0	0	Cobbles	Pebbles	Gravel	0	None	0	1	0	grassland
03-Aug-05	Briggle	357227	534644	0	7	7	2	2	0	Boulders	Gravel	Silt/mud	0	None	0	1	0	grassland
03-Aug-05	Briggle	358838	533549	2	10	12	5	0	0	Pebbles	Cobbles	Gravel	0	None	0	1	0	Woodland
03-Aug-05	Briggle	360688	532672	3	0	3	0	0	0	Cobbles	Boulders	Pebbles	1	Undercut	0	1	0	Woodland
11-Aug-05	Caldbeck	329958	535140	5	0	5	3	2	0	Cobbles	Pebbles	Gravel	0	Earth cliff	1	0	0	grassland
11-Aug-05	Caldbeck	330062	535944	0	0	0	0	1	0	Boulders	Cobbles	Pebbles	0	Undercut	1	0	0	grassland
11-Aug-05	Caldbeck	330885	539668	0	0	0	0	3	0	Cobbles	Boulders	Pebbles	0	None	0	1	0	grassland
11-Aug-05	Caldbeck	333210	539839	2	4	6	4	0	0	Cobbles	Boulders	Pebbles	0	Poached	1	0	0	grassland
09-Aug-05	Caldew	332166	532190	0	1	1	3	0	3	Boulders	Cobbles	Pebbles	0	Undercut	0	0	0	grassland
09-Aug-05	Caldew	333260	532664	1	1	2	1	0	6	Boulders	Cobbles	Pebbles	0	None	0	0	0	grassland
09-Aug-05	Caldew	331374	531378	0	13	13	6	1	2	Cobbles	Boulders	Pebbles	0	Undercut	1	0	0	grassland
09-Aug-05	Caldew	334676	531945	0	5	5	3	0	2	Cobbles	Pebbles	Gravel	0	Undercut	0	0	2	grassland
09-Aug-05	Caldew	336577	532544	0	9	9	3	0	0	Cobbles	Pebbles	Gravel	0	None	0	1	0	grassland
11-Aug-05	Caldew	335092	540215	0	11	11	4	0	2	Boulders	Cobbles	Pebbles	0	None	0	0	0	Woodland
11-Aug-05	Caldew	335817	541687	0	2	2	0	0	4	Boulders	Cobbles	Pebbles	0	None	0	0	0	Woodland
12-Aug-05	Caldew	336342	533376	0	66	66	20	0	0	Cobbles	Pebbles	Gravel	0	None	0	0	1	grassland
12-Aug-05	Caldew	336038	534217	1	30	31	8	1	0	Pebbles	Gravel	Sand	0	Poached	1	0	0	grassland
12-Aug-05	Caldew	336315	535215	0	40	40	10	0	2	Boulders	Cobbles	Pebbles	0	None	1	1	0	grassland
12-Aug-05	Caldew	336769	536566	0	10	10	4	0	2	Boulders	Cobbles	Pebbles	0	None	0	1	0	Woodland
12-Aug-05	Caldew	336188	537746	0	0	0	0	0	1	Cobbles	Boulders	Pebbles	0	None	0	1	0	grassland
08-Aug-05	Croglin	354694	544342	0	0	0	0	1	0	Boulders	Cobbles	Pebbles	0	Poached	1	0	0	grassland
08-Aug-05	Croglin	355811	545820	3	0	3	1	3	0	Boulders	Cobbles	Pebbles	0	Poached	1	0	0	grassland
08-Aug-05	Croglin	355208	545239	0	0	0	0	5	0	Boulders	Cobbles	Pebbles	0	None	1	0	0	grassland
12-Sep-05	Croglin	353923	42808	1	0	1	0	0	0	Cobbles	Pebbles	Gravel	0	None	0	0	10	Woodland
30-Aug-05	Dacre	344715	526357	0	11	11	3	1	1	Boulders	Cobbles	Pebbles	1	None	1	1	0	grassland
30-Aug-05	Dacre	347709	526757	1	20	21	6	0	4	Cobbles	Pebbles		0	None	1	0	0	grassland
30-Aug-05	Dacre	346069	526282	0	9	9	3	0	3	Bedrock	Boulders	Cobbles	0	None	0	1	0	grassland
30-Aug-05	Dacre	345328	526078	0	11	11	3	0	4	Bedrock	Boulders	Cobbles	0	None	0	0	0	Woodland

2005 Survey: Semi-quantitative electrofishing with habitat data (Eden Rivers Trust)

Date of survey	Site	Easting	Northing	0+ trout	0+ salmon	Total 0+ salmon & trout	0+ salmon and trout missed	1+ and older trout	1+ and older salmon	Dominant substrate	Substrate 2	Substrate 3	Siltation 1=present 0=absent	Erosion type	Stock access 1=Yes 0=No	Tunnelling 1=present 0=absent	Fence width (m)	Landuse
30-Aug-05	Dacre (Skirwith)	342116	526942	4	0	4	2	8	2	Boulders	Cobbles	Pebbles	0	None	0	0	0	grassland
30-Aug-05	(Thackthwaite)	342073	525913	5	43	48	6	0	1	Cobbles	Pebbles	Gravel	0	None	0	1	0	grassland
30-Aug-05	(Thackthwaite)	341166	525284	6	26	32	6	0	3	Gravel	Pebbles	Sand	1	None	1	0	0	grassland
02-Sep-05	Deepdale	338723	512924	3	0	3	1	1	0	Boulders	Cobbles	Pebbles	0	None	1	0	0	grassland
02-Sep-05	Deepdale	339147	513439	6	0	6	2	0	0	Boulders	Cobbles	Pebbles	0	None	1	0	0	grassland
02-Sep-05	Deepdale	339629	514075	2	0	2	1	0	0	Bedrock	Boulders	Cobbles	0	None	1	0	0	grassland
02-Sep-05	Deepdale	339940	514347	2	15	17	3	0	1	Boulders	Bedrock	Cobbles	0	Undercut	1	1	0	grassland
04-Aug-05	Eden	356728	533487	0	7	7	2	0	2	Boulders	Cobbles	Pebbles	0	None	0	0	0	Arable crop
01-Aug-05	Eden	373568	515919	0	61	61	15	1	3	Pebbles	Cobbles	Gravel	0	Earth cliff	0	0	5	grassland
27-Jul-05	Eden	361214	525831	0	23	23	8	0	0	Pebbles	Cobbles	Gravel	0	Poached	1	0	0	grassland
10-Aug-05	Eden	351538	548341	0	37	37	20	0	0	Cobbles	Boulders	Pebbles	0	None	0	0	0	grassland
10-Aug-05	Eden	348271	552273	0	4	4	2	0	0	Cobbles	Boulders	Pebbles	0	Earth cliff	0	0	0	grassland
10-Aug-05	Eden	346762	555413	0	3	3	2	0	0	Cobbles	Pebbles	Gravel	0	None	0	0	0	grassland
10-Aug-05	Eden	346977	556693	0	10	10	6	0	0	Cobbles	Pebbles		0	Poached	1	0	0	grassland
10-Aug-05	Eden	346706	557959	0	17	17	6	0	0	Cobbles	Pebbles	Gravel	0	None	0	0	5	grassland
10-Aug-05	Eden	344164	555098	0	1	1	3	0	0	Pebbles	Gravel	Cobbles	0	Poached	1	0	0	grassland
12-Sep-05	Eden	355100	540372	0	5	5	2	0	0	Cobbles	Boulders	Pebbles	0	Poached	1	0	0	grassland
05-Aug-05	Glassonby	358942	539706	8	5	13	4	3	0	Cobbles	Pebbles	Gravel	0	None	0	1	0	Woodland
05-Aug-05	Glassonby	357857	539403	4	0	4	1	1	1	Cobbles	Boulders	Pebbles	0	None	0	1	0	Woodland
05-Aug-05	Glassonby beck	356590	539557	0	49	49	6	0	7	Cobbles	Boulders	Pebbles	0	Poached	1	0	0	grassland
26-Aug-05	Glencoyne	337991	518639	6	0	6	4	2	0	Pebbles	Cobbles	Boulders	0	None	1	0	0	grassland
26-Aug-05	Glencoyne	338224	518704	0	0	0	0	1	0	Boulders	Cobbles	Pebbles	0	None	0	0	0	grassland
26-Aug-05	Glencoyne	338564	518722	5	0	5	2	0	0	Cobbles	Boulders	Pebbles	0	Undercut	0	0	5	grassland
26-Aug-05	Glenridding	337990	516934	0	0	0	2	1	0	Boulders	Cobbles	Pebbles	0	None	0	0	0	grassland
26-Aug-05	Glenridding	338621	516890	2	0	2	1	0	0	Cobbles	Boulders	Pebbles	0	None	0	0	0	Woodland
26-Aug-05	Glenridding	338828	516951	2	1	3	1	0	0	Cobbles	Boulders	Pebbles	0	None	0	0	0	Park/garden
26-Aug-05	Glenridding	336706	517356	0	0	0	0	0	0	Boulders	Bedrock	Cobbles	0	None	0	0	15	grassland
26-Aug-05	Glenridding	337808	517008	2	0	2	0	1	0	Boulders	Cobbles	Pebbles	0	None	0	0	0	Park/garden
02-Sep-05	Goldril	340403	514755	0	6	6	0	0	0	Pebbles	Cobbles	Gravel	0	Undercut	0	0	0	grassland
02-Sep-05	Goldrill	340254	513407	1	1	2	1	0	0	Cobbles	Pebbles	Gravel	0	None	0	0	0	Woodland
02-Sep-05	Goldrill	339639	516098	0	4	4	2	0	0	Pebbles	Gravel	Sand	0	None	0	0	1	grassland
16-Aug-05	Grisedale	336284	514375	5	0	5	1	0	7	Bedrock	Boulders	Cobbles	0	None	1	0	0	grassland
16-Aug-05	Grisedale	336843	514963	2	0	2	1	2	1	Cobbles	Pebbles	Gravel	0	None	1	0	0	grassland
16-Aug-05	Grisedale	337646	515300	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	0	None	0	0	2	grassland
16-Aug-05	Grisedale	338146	515693	2	0	2	1	0	0	Cobbles	Boulders	Pebbles	0	None	0	0	2	grassland
16-Aug-05	Grisedale	338920	516100	3	4	7	2	0	4	Boulders	Cobbles	Pebbles	0	None	0	1	0	Woodland
16-Aug-05	Grisedale	339128	516195	0	10	10	4	0	2	Cobbles	Pebbles	Gravel	0	None	0	1	0	Park/garden
01-Aug-05	Hartley	377661	509014	22	11	33	8	4	4	Cobbles	Pebbles	Gravel	0	None	0	0	12	Woodland
01-Aug-05	Hartley	378188	508987	15	0	15	5	4	0	Cobbles	Pebbles	Gravel	0	None	0	1	0	Woodland

2005 Survey: Semi-quantitative electrofishing with habitat data (Eden Rivers Trust)

Date of survey	Site	Easting	Northing	0+ trout	0+ salmon	Total 0+ salmon & trout	0+ salmon and trout missed	1+ and older trout	1+ and older salmon	Dominant substrate	Substrate 2	Substrate 3	Siltation 1=present 0=absent	Erosion type	Stock access 1=Yes 0=No	Tunnelling 1=present 0=absent	Fence width (m)	Landuse
26-Jul-05	Haweswater	352000	516582	0	11	11	5	0	0	Cobbles	Pebbles	Gravel	0	None	1	0	0	grassland
01-Sep-05	Hayeswater	341750	512810	0	0	0	0	0	0	Boulders	Cobbles	Pebbles	0	None	1	0	0	grassland
01-Sep-05	Hayeswater	340890	513000	2	0	2	1	0	0	Cobbles	Boulders	Pebbles	0	None	0	0	0	grassland
01-Sep-05	Hayeswater	340580	513320	0	2	2	1	0	2	Cobbles	Pebbles	Gravel	0	None	0	1	0	grassland
26-Jul-05	Heltondale	350347	520814	1	3	4	0	4	4	Boulders	Cobbles	Pebbles	0	None	0	0	0	grassland
26-Jul-05	Heltondale	350690	520589	1	5	6	3	0	2	Cobbles	Boulders	Pebbles	0	None	1	0	0	grassland
01-Aug-05	Hilton	373117	520732	0	8	8	3	2	20	Bedrock	Bedrock	Cobbles	0	None	0	0	0	grassland
01-Aug-05	Hilton	372009	520679	2	2	4	2	6	0	Boulders	Bedrock	Cobbles	0	Undercut	0	0	0	grassland
12-Jul-05	Hilton	373851	521042	1	0	1	0	1	3	Boulders	Cobbles	Pebbles	0	None	1	0	0	grassland
01-Aug-05	Hilton- Coupland	370923	518821	2	40	42	6	0	0	Cobbles	Pebbles	Gravel	0	None	0	0	8	grassland
26-Jul-05	Howe	351665	518131	3	2	5	2	0	1	Cobbles	Pebbles	Gravel	0	None	0	1	0	grassland
26-Jul-05	Howe	350687	517666	0	0	0	0	3	0	Gravel	Pebbles	Cobbles	0	None	1	0	0	grassland
15-Aug-05	Howgrain	343374	518170	11	28	39	7	2	1	Cobbles	Pebbles		0	Poached	1	0	0	grassland
03-Aug-05	Kirkland (Briggle)	364484	532410	27	0	27	15	5	0	Boulders	Cobbles	Pebbles	1	Undercut	1	0	0	grassland
03-Aug-05	Kirkland (Briggle)	365322	532762	15	0	15	6	0	0	Bedrock	Boulders	Cobbles	1	Undercut	1	0	0	grassland
01-Sep-05	Kirkstone	339950	511450	2	3	5	2	0	0	Cobbles	Pebbles	Gravel	0	Earth cliff	0	0	1	grassland
01-Sep-05	Kirkstone (Dovedale)	339820	511780	0	6	6	2	0	0	Cobbles	Pebbles	Gravel	0	None	0	0	4	grassland
22-Aug-05	Leith	357982	524604	0	18	18	5	0	0	Gravel	Cobbles	Pebbles	1	Undercut	1	1	0	grassland
22-Aug-05	Leith	355077	523870	0	0	0	0	0	0	Bedrock	Cobbles	Gravel	0	Undercut	1	1	0	grassland
22-Aug-05	Leith	355505	524777	0	3	3	0	0	1	Cobbles	Boulders	Pebbles	0	None	0	1	0	grassland
22-Aug-05	Leith	356325	525110	0	13	13	3	0	1	Cobbles	Pebbles	Gravel	0	None	0	0	0	grassland
01-Sep-05	Leith	356624	525211	0	5	5	1	0	0	Cobbles	Boulders	Gravel	1	Earth cliff	1	0	0	grassland
01-Sep-05	Leith	355591	518333	1	0	1	0	3	0	Cobbles	Pebbles	Gravel	0	None	0	0	0	Woodland
01-Sep-05	Leith	355876	518655	0	0	0	0	9	0	Pebbles	Cobbles	Gravel	1	Poached	1	0	0	grassland
31-Aug-05	Leith	355779	519889	2	0	2	0	4	0	Boulders	Bedrock	Cobbles	1	Poached	1	0	0	grassland
31-Aug-05	Leith	355729	520366	2	0	2	1	3	0	Cobbles	Pebbles	Boulders	1	Poached	1	0	0	grassland
31-Aug-05	Leith	355809	520683	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	1	Poached	1	0	0	grassland
31-Aug-05	Leith	355167	522447	0	0	0	0	2	0	Cobbles	Boulders	Pebbles	0	None	0	1	0	Park/garden
31-Aug-05	Leith	355146	523142	0	0	0	0	3	0	Cobbles	Pebbles	Boulders	1	Poached	1	0	0	grassland
13-Sep-05	Leith	359009	524470	0	5	5	1	0	1	Cobbles	Pebbles	Gravel	1	Poached	1	0	0	grassland
04-Aug-05	Lowther	352667	528651	0	3	3	2	1	5	Boulders	Cobbles	Pebbles	0	None	0	0	5	grassland
29-Jul-05	Lowther	355990	513747	0	69	69	30	3	0	Pebbles	Cobbles	Gravel	0	None	1	0	0	grassland
29-Jul-05	Lowther	355318	513540	0	0	0	0	0	0	Cobbles	Boulders		0	None	1	0	0	grassland
29-Jul-05	Lowther	352084	518008	0	45	45	20	0	0	Cobbles	Pebbles	Gravel	0	None	0	0	0	Park/garden
29-Jul-05	Lowther	351738	521392	0	30	30	5	2	0	Pebbles	Gravel		0	None	1	0	0	grassland
29-Jul-05	Lowther	351822	522212	0	43	43	12	0	3	Pebbles	Cobbles	Gravel	0	None	1	0	0	grassland
29-Jul-05	Lowther	352036	525011	0	2	2	1	0	2	Boulders	Cobbles		0	None	1	0	0	grassland
28-Jul-05	Lowther	355999	512641	0	18	18	8	0	0	Boulders	Cobbles		0	Undercut	1	0	0	grassland
28-Jul-05	Lowther	353390	516783	0	32	32	12	0	0	Cobbles	Pebbles	Gravel	0	None	0	0	0	Woodland

2005 Survey: Semi-quantitative electrofishing with habitat data (Eden Rivers Trust)

Date of survey	Site	Easting	Northing	0+ trout	0+ salmon	Total 0+ salmon & trout	0+ salmon and trout missed	1+ and older trout	1+ and older salmon	Dominant substrate	Substrate 2	Substrate 3	Siltation 1=present 0=absent	Erosion type	Stock access 1=Yes 0=No	Tunnelling 1=present 0=absent	Fence width (m)	Landuse
29-Jul-05	Lowther (knipe)	351523	519784	0	28	28	10	2	0	Gravel	Pebbles		0	None	1	0	0	grassland
29-Jul-05	Lowther (swindale)	351510	513201	0	4	4	2	0	7	Cobbles	Boulders	Pebbles	0	Undercut	1	0	0	sh
29-Jul-05	Lowther (Swindale)	352248	514442	2	45	47	8	0	3	Pebbles	Cobbles	Gravel	0	Undercut	0	0	0	grassland
26-Jul-05	Lowther (swindale)	350657	512297	0	15	15	7	0	2	Pebbles	Cobbles	Gravel	0	None	1	0	0	grassland
28-Jul-05	Lowther.	355155	514298	0	75	75	40	0	0	Cobbles	Pebbles	Gravel	0	None	1	0	0	grassland
17-Jul-05	Lyvennet	360837	525928	0	34	34	4	0	0	Cobbles	Pebbles	Gravel	0	None	0	0	10	grassland
27-Jul-05	Lyvennet	360582	524070	0	5	5	2	0	1	Bedrock	Cobbles	Pebbles	0	None	0	1	0	grassland
25-Jul-05	Lyvennet	361939	520145	0	17	17	6	0	0	Cobbles	Pebbles	Gravel	0	None	0	0	0	grassland
25-Jul-05	Lyvennet	361896	520235	0	17	17	5	0	0	Cobbles	Pebbles	Boulders	0	None	0	0	5	grassland
25-Jul-05	Lyvennet	361776	521454	0	16	16	4	2	0	Boulders	Bedrock	Pebbles	0	None	0	1	0	Woodland
25-Jul-05	Lyvennet	362270	515083	2	0	2	1	3	0	Cobbles	Boulders	Pebbles	0	Poached	1	1	0	grassland
25-Jul-05	Lyvennet	362302	514434	3	0	3	1	0	0	Cobbles	Boulders		0	None	1	0	0	grassland
25-Jul-05	Lyvennet	362188	518139	1	20	21	6	0	0	Pebbles	Boulders	Cobbles	0	Earth cliff	0	0	3	grassland
04-Aug-05	Lyvennet	360990	524862	0	20	20	6	0	5	Cobbles	Boulders	Pebbles	0	None	0	0	10	Arable crop
27-Jul-05	Morland	360664	523906	0	4	4	2	0	0	Boulders	Cobbles	Pebbles	1	None	1	0	0	grassland
15-Aug-05	Rampsgill	343822	516365	18	20	38	12	1	2	Cobbles	Pebbles	Gravel	0	Poached	1	0	0	grassland
05-Aug-05	Raven beck	360844	543235	2	0	2	1	5	0	Bedrock	Boulders	Cobbles	0	Earth cliff	0	0	0	Woodland
05-Aug-05	Raven beck	359234	542596	1	0	1	0	7	0	Boulders	Cobbles	Pebbles	0	None	0	1	0	grassland
05-Aug-05	Raven beck	360205	543045	0	0	0	0	5	0	Boulders	Cobbles		0	None	0	1	0	Woodland
08-Aug-05	Raven beck	355233	541069	0	21	21	6	0	8	Cobbles	Pebbles	Gravel	0	None	0	1	0	Park/garden
05-Aug-05	Robbery water	356618	535947	0	167	167	35	1	3	Cobbles	Pebbles	Gravel	0	Poached	1	0	0	grassland
08-Aug-05	Robbery water	357866	535439	8	31	39	6	3	7	Cobbles	Boulders	Pebbles	1	None	1	1	0	grassland
08-Aug-05	Robbery water	359210	536083	9	26	35	4	3	0	Pebbles	Cobbles	Sand	0	None	0	1	0	grassland
08-Aug-05	Robbery water	359847	537322	16	2	18	1	5	0	Cobbles	Pebbles	Gravel	0	Poached	1	1	0	grassland
15-Aug-05	Sandwick	342515	519415	5	15	20	4	1	3	Boulders	Cobbles	Pebbles	0	None	1	1	0	grassland
05-Sep-05	Scandal	374309	509888	0	22	22	6	2	1	Boulders	Cobbles	Pebbles	0	Earth cliff	0	0	0	grassland
05-Sep-05	Scandal	373077	508242	0	7	7	3	0	0	Boulders	Cobbles	Pebbles	0	None	0	0	0	grassland
05-Sep-05	Scandal	376456	511216	0	23	23	6	0	0	Cobbles	Pebbles	Gravel	0	Earth cliff	1	0	0	grassland
05-Sep-05	Scandal	375391	511032	0	10	10	3	0	0	Cobbles	Pebbles		0	Earth cliff	1	0	0	grassland
05-Sep-05	Scandal	374745	510419	0	11	11	6	3	6	Cobbles	Boulders		0	Earth cliff	0	0	2	grassland
05-Sep-05	Scandal	372755	506754	0	2	2	1	3	7	Cobbles	Boulders	Pebbles	0	None	0	0	5	grassland
05-Sep-05	Scandal	371932	505086	0	4	4	2	0	4	Boulders	Bedrock	Cobbles	0	Undercut	0	0	5	grassland
02-Aug-05	Swindale	377422	513448	0	37	37	8	0	2	Cobbles	Pebbles	Gravel	0	Earth cliff	1	0	0	grassland
02-Aug-05	Swindale	348160	513903	0	19	19	4	0	5	Cobbles	Boulders	Pebbles	0	Earth cliff	1	0	0	grassland
01-Aug-05	Swindale	379260	514176	0	20	20	8	0	1	Boulders	Cobbles	Pebbles	0	Poached	1	0	0	grassland

2005 Quantitative cluster electrofishing survey of the Upper River Eden (Eden Rivers Trust)

Date of survey	Site	Easting	Northing	Channel length surveyed (m)	Mean channel width (m)	% Riffle	% Run	% Glide or Pool	Total 0+ Trout	Total 0+ Salmon	Total 1+ and older Trout	Total 1+ and older Salmon	Pass 1 0+ Trout	Pass 1 0+ Salmon	Pass 1 1+ Trout	Pass 1 1+ Salmon	Pass 2 0+ Trout	Pass 2 0+ Salmon	Pass 2 1+ Trout	Pass 2 1+ Salmon	Pass 3 0+ Trout	Pass 3 0+ Salmon	Pass 3 1+ Trout	Pass 3 1+ Salmon	Dominant substrate	Substrate 2	Substrate 3	Bank Profile	Erosion Type	Stock access 1=yes 0=no	Fence width (m)	Tunnelling 1=present 0=absent	Land Use
20/07/2005	Q1 Outhgill Upper Eden	378258	501556	33	2	90	0	10	0	0	14	0	0	0	11	0	0	0	2	0	0	0	1	0	Cobbles	Boulders	Pebbles	Steep		0	0	0	Unimproved grassland
20/07/2005	C1 Deep Gill Upper Eden	378160	500494	35	1	60	0	40	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	Cobbles	Boulders	Pebbles		None	1	0	0	Unimproved grassland
20/07/2005	Q2 Upper Eden	378281	503924	34	8	85	0	15	18	0	5	0	10	0	2	0	6	0	2	0	2	0	1	0	Cobbles	Boulders	Pebbles		Earth cliff	0	0	0	Unimproved grassland
20/07/2005	C2b Upper Eden	378188	500510	49	6	70	0	30	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	Boulders	Cobbles	Pebbles	Shallow	None	1	0	0	Unimproved grassland
20/07/2005	C2a Upper Eden	377702	504491	70	9	75	0	25	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	Cobbles	Pebbles	Boulders	Shallow	Poached	1	0	0	Unimproved grassland
21/07/2005	Q3 Upper Eden	378894	510444	32	10	40	0	60	3	118	4	19	2	67	3	14	1	31	1	3	0	18	0	3	Bedrock	Boulders	Cobbles	Steep	None	0	3	0	Improved grassland
21/07/2005	C2c Upper Eden	377211	505858	50	7	20	30	50	4	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	Cobbles	Boulders	Bedrock	Steep	None	0	3	0	Woodland
21/07/2005	C3a Upper Eden	377747	507988	34	7	45	0	55	4	29	0	7	4	29	0	7	0	0	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	Steep	None	1	0	0	Improved grassland
21/07/2005	C3b Upper Eden	376963	512813	30	8	40	50	10	0	65	1	1	0	65	1	1	0	0	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	Shallow	Earth cliff	0	8	1	Improved grassland
17/07/2005	Q4 Helm	371354	510978	27	3	60	0	40	3	37	1	0	3	17	1	0	0	7	0	0	0	5	0	0	Cobbles	Pebbles	Gravel	Steep	None	1	0	0	Improved grassland
16/07/2005	C4c Helm	370902	510034	57	3	80	0	20	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	Shallow	None	1	0	0	Improved grassland
16/07/2005	C4b Helm	371024	510397	40	3	40	0	60	10	4	4	0	10	4	4	0	0	0	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	Shallow	Earth cliff	1	0	0	Unimproved grassland
16/07/2005	C4a Helm	371318	512277	46	4	25	0	75	0	5	2	0	0	5	2	0	0	0	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	Shallow	Poached	1	0	0	Improved grassland
16/07/2005	Q5 Helm	370161	516577	38	5	60	0	40	0	6	3	3	0	3	1	1	0	2	1	2	0	1	1	0	Cobbles	Boulders	Pebbles	Shallow	None	0	0	0	Woodland
16/07/2005	C5a Helm	370173	516424	45	2	25	0	75	0	2	0	1	0	2	0	1	0	0	0	0	0	0	0	0	Cobbles	Pebbles		Steep	Poached	1	0	0	Improved grassland
19/07/2005	Q10 Lyvennet	361257	523281	42	10	65	0	35	0	139	0	0	0	90	0	0	0	34	0	0	0	15	0	0	Pebbles	Cobbles	Gravel	Steep	None	0	5	0	Improved grassland
19/07/2005	C10b Lyvennet	360019	524786	42	7	70	0	30	0	23	0	0	0	23	0	0	0	0	0	0	0	0	0	0	Bedrock	Cobbles	Pebbles	Steep	None	0	5	0	Improved grassland
19/07/2005	C10a Lyvennet	361448	521967	30	9	50	0	50	0	3	0	0	0	3	0	0	0	0	0	0	0	0	0	0	Bedrock	Cobbles		Steep	None	0	0	0	Woodland
23/07/2005	C10c Lyvennet	360157	524520	58	5	60	20	20	0	32	1	2	0	32	1	2	0	0	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	Steep	None	0	6	0	Improved grassland
11/07/2005	Q11 Trout	365326	524342	42	6	40	20	40	1	60	7	3	0	56	6	2	1	15	1	1	0	9	0	0	Pebbles	Cobbles	Gravel	Steep	None	0	4	1	Improved grassland
11/07/2005	C11a Trout	366133	524077	45	7	70	0	30	1	15	3	3	1	15	3	3	0	0	0	0	0	0	0	0	Pebbles	Gravel	Sand	Shallow	Earth cliff	1	0	0	Improved grassland
11/07/2005	C11b Trout	366770	524364	34	5	20	0	80	0	6	1	5	0	6	1	5	0	0	0	0	0	0	0	0	Cobbles	Gravel	Sand	Steep	Poached	1	0	0	Improved grassland
09/07/2005	Q12 Trout	369723	523185	30	3	80	15	5	1	22	15	14	1	16	10	9	0	5	3	2	0	1	2	3	Boulders	Pebbles	Cobbles	Shallow	None	0	0	0	Woodland
09/07/2005	C12a Trout	369113	523192	37	4	70	0	30	2	13	10	5	2	13	10	5	0	0	0	0	0	0	0	0	Boulders	Bedrock	Cobbles	Steep	None	0	0	0	Woodland
09/07/2005	C12b Trout	370769	521650	36	2	50	0	50	0	0	2	0	0	0	2	0	0	0	0	0	0	0	0	0	Silt/mud	Pebbles	Gravel	Shallow	None	0	0	0	Woodland
10/07/2005	C12c Trout	368159	523410	44	5	60	0	40	0	30	4	2	0	30	4	2	0	0	0	0	0	0	0	0	Gravel	Pebbles	Silt/mud	Steep	None	0	0	1	Woodland
10/07/2005	Q13 Trout	368419	526948	37	2	95	0	5	9	0	5	11	6	0	2	8	1	0	2	1	2	0	1	2	Boulders	Cobbles	Pebbles	Steep	Poached	1	2	0	Improved grassland
16/07/2005	C5b Helm	370916	516920	46	4	30	0	70	0	3	3	0	0	3	3	0	0	0	0	0	0	0	0	0	Cobbles	Pebbles		Shallow	Earth cliff	0	2	0	Improved grassland
13/07/2005	Q6 Asby/Hoff	368858	513773	57	4	45	0	55	24	0	7	0	15	0	4	0	5	0	2	0	4	0	1	0	Cobbles	Pebbles	Gravel	Shallow	Poached	1	0	0	Improved grassland
13/07/2005	C6b Asby/Hoff	368276	513361	33	4	70	0	30	2	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	Steep	None	0	0	0	Park/garden
13/07/2005	C6a Asby/Hoff	368809	514686	60	4	30	0	70	1	0	2	0	1	0	2	0	0	0	0	0	0	0	0	0	Cobbles	Pebbles		Shallow	Poached	1	0	0	Improved grassland
17/07/2005	Q7 Hoff	367264	514506	32	3	20	0	80	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	Cobbles	Pebbles		Steep	Earth cliff	0	0	0	Improved grassland
17/07/2005	C7a Hoff	366374	514099	35	2	20	0	80	0	0	2	0	0	0	2	0	0	0	0	0	0	0	0	0	Cobbles	Pebbles	Boulders	Steep	Earth cliff	1	0	0	Unimproved grassland
17/07/2005	C7b Hoff	367869	515116	35	4	50	0	50	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	Steep	None	0	1	0	Improved grassland
12/07/2005	Q8 Hilton	374331	521047	34	4	80	0	20	3	0	7	18	0	0	3	10	3	0	2	5	0	0	2	3	Boulders	Cobbles	Pebbles	Shallow	None	1	0	0	Unimproved grassland
12/07/2005	C8a Hilton	375910	522227	27	4	80	0	20	0	0	10	4	0	0	10	4	0	0	0	0	0	0	0	0	Boulders	Cobbles	Pebbles	Shallow	None	1	0	0	Unimproved grassland
12/07/2005	C8b Hilton	371046	520052	47	5	95	0	5	0	7	1	10	0	7	1	10	0	0	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	Shallow	None	1	0	0	Unimproved grassland
18/07/2005	Q9 Lyvennet	362557	516172	50	8	90	0	10	12	87	0	0	8	51	0	0	3	25	0	0	1	11	0	0	Cobbles	Pebbles		Shallow	None	1	0	0	Unimproved grassland
18/07/2005	C9d Lyvennet	362473	517072	38	8	25	0	75	1	1	0	0	1	1	0	0	0	0	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	Shallow	None	1	0	0	Improved grassland
18/07/2005	C9c Lyvennet	362282	517355	47	3	30	10	60	0	6	2	1	0	6	2	1	0	0	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	Steep	Poached	1	0	0	Improved grassland .
18/07/2005	C9b Lyvennet	362046	519003	30	9	60	0	40	0	50	1	2	0	50	1	2	0	0	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	Steep	None	0	0	1	Improved grassland
19/07/2005	C9a Lyvennet	362171	518740	38	12	75	0	25	0	59	0	2	0	59	0	2	0	0	0	0	0	0	0	0	Bedrock	Cobbles	Pebbles	Shallow	None	0	0	0	Unimproved grassland
10/07/2005	C13 Trout	368593	525404	44	4	45	0	55	1	15	1	6	1	15	1	6	0	0	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	Steep	Poached	1	0	1	Improved grassland
15/07/2005	Q14 Crowdundle	384489	530533	49	4	70	15	15	25	123	9	18	14	80	5	9	6	29	3	6	5	14	1	3	Boulders	Cobbles		Shallow	Poached	1	3	0	Improved grassland
15/07/2005	C14a Crowdundle	384347	530090	41	4	70	0	30	7	25	4	6	7	25	4	6	0	0	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	Steep	Earth cliff	0	2	0	Improved grassland
15/07/2005	C5c Helm	370503	514188	42	5	45	0	55	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	Shallow	Poached	1	0	0	Improved grassland
14/07/2005	C14b Crowdundle	363019	529278	39	4	25	25	50	5	14	2	4	5	14	2	4	0	0	0	0	0	0	0	0	Boulders	Bedrock	Cobbles	Steep	None	0	0	1	Woodland
14/07/2005	Q15 Crowdundle	361169	528158	45	6	30	0	70	0	82	6	9	0	64	3	6	0	12	1	2	0	6	2	1	Pebbles	Cobbles	Gravel	Steep	None	0	0	1	Improved grassland
14/07/2005	C15a Crowdundle	362138	528407	42	5	70	20	10	2	58	3	5	2	58	3	5	0	0	0	0	0	0	0	0	Cobbles	Bedrock	Pebbles	Steep	Earth cliff	1	0	0	Improved grassland
14/07/2005	C15b Crowdundle	360791	528205	45	7	60	5	35	1	107	4	8	1	107	4	8	0	0	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	Steep	None	0	2	0	Improved grassland
22/07/2005	Q16 Aira beck	337152	520768	30	4	60	0	40	0	0	10	0	0	0	5	0	0	0	2	0	0	0	3	0	Boulders	Cobbles	Pebbles	Shallow	Undercut	1	0	0	Unimproved grassland
22/07/2005	C16b Aira beck	338318	521444	26	4	60	0	40	0	0	8	0	0	0	8	0	0	0	0	0	0	0	0	0	Boulders	Cobbles	Pebbles	Steep	None	1	1	0	Unimproved grassland
22/07/2005	C16c Aira beck	339631	521614	35	5	0	50	50	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	Bedrock	Boulders	Cobbles	Steep	None	1	0	0	Unimproved grassland
22/07/2005	C16a Aira beck	339326	521582	35	4	60	0	40	0	0	13	0	0	0	13	0	0	0	0	0	0	0	0	0	Cobbles	Bedrock	Pebbles	Steep	None	0	0	1	Woodland

2006 Survey: Semi-quantitative electrofishing with habitat data (Eden Rivers Trust)

							No of fish caugh in 5 minutes of fishing															
Date of survey	Site	Sub-catchment	Easting	Northing	Altitude (m above sea level)	Water temp °C	0+ Trout	0+ Salmon	Total 0+ salmon and trout	1+ and older Trout	1+ and older Salmon	No. of 0+ fish missed	Dominant substrate	Substrate 2	Substrate 3	Siltation 1=present 0=absent	River width (m)	Erosion type	Stock access 1=yes 0=no	Fence width (m)	% Channel Cover	Landuse
17/07/2006	Hartley	Eden	378300	508690	0	14	37	0	37	2	0	10	Gravel	Pebbles	Cobbles	0	2	None	0	0	80	Park/garden
17/07/2006	Hartley	Eden	377700	509100	0	21	27	6	33	2	2	10	Pebbles	Cobbles	Gravel	0	2	None	0	10	60	Improved grassland
17/07/2006	Ladthwaite	Eden	379700	506800	0	14	7	0	7	3	0	3	Cobbles	Boulders	Pebbles	0	3	None	1	3	5	Improved grassland
17/07/2006	Unnamed off Ladtwaite	Ladthwaite	379950	506650	0	14	3	0	3	2	0	1	Cobbles	Pebbles		1	1	None	1	0	0	Improved grassland
17/07/2006	Ladtwaite	Eden	380000	506700	0	21	0	0	0	2	0	0	Boulders	Cobbles	Pebbles	0	3	Undercut	1	0	5	Unimproved grassland
17/07/2006	Ais Gill	Upper Eden	377400	497550	0	15	3	0	3	2	0	1	Cobbles	Bedrock	Bedrock	0	3	None	0	2	5	Improved grassland
17/07/2006	Outhgill	Upper Eden	378290	501550	0	18	3	0	3	8	0	1	Cobbles	Pebbles	Gravel	0	2	None	0	1	40	Improved grassland
17/07/2006	Aller beck	Upper Eden	378510	503390	0	13	0	0	0	14	0	0	Cobbles	Pebbles	Gravel	0	1	Poached	1	0	15	Improved grassland
18/07/2006	Sweetmilk sike	Upper Eden	384750	521200	0	15	0	0	0	0	0	0	Cobbles	Pebbles	Silt/mud	1	2	Poached	1	0	50	Improved grassland
18/07/2006	Sweetmilk sike	Upper Eden	385200	521750	0	16	0	0	0	0	0	0	Cobbles	Pebbles	Silt/mud	1	3	Poached	0	2	90	Improved grassland
18/07/2006	Lowgill beck	Upper Eden	376210	515100	0	19	0	0	0	1	0	0	Gravel	Pebbles	Sand	1	1	None	0	2	0	Improved grassland
18/07/2006	Greenber Sike	Upper Eden	371325	516367	168	27	0	0	0	0	0	0	Silt/mud	Sand	Gravel	1	1	Poached	1	0	0	Improved grassland
20/07/2006	Rudhills Eden	Upper Eden	376908	513238	147	19	0	103	103	0	0	30	Cobbles	Pebbles	Gravel	0	12	Earth cliff	0	2	0	Improved grassland
20/07/2006	Woodhouse Eden	Upper Eden	375798	513642	142	19	0	157	157	0	0	35	Cobbles	Pebbles	Gravel	0	10	Earth cliff	1	1	0	Improved grassland
20/07/2006	Ormside Eden	Upper Eden	370128	517747	127	18	0	29	29	0	0	8	Pebbles	Cobbles	Gravel	0	15	None	0	5	0	Improved grassland
21/07/2006	Hoff	Hoff	366950	519650	0	18	0	15	15	0	1	6	Boulders	Cobbles	Pebbles	0	7	None	0	4	60	Improved grassland
21/07/2006	Nether Hoff Sike	Hoff	366750	520000	0	19	0	0	0	0	0	0	Cobbles	Sand	Silt/mud	1	2	Poached	1	0	30	Improved grassland
21/07/2006	Hoff Row Hoff	Hoff	367350	518500	0	22	6	10	16	0	0	3	Cobbles	Boulders	Gravel	0	6	None	0	0	0	Improved grassland
21/07/2006	Oakbeck	Hoff	367400	516780	0	21	0	0	0	1	0	0	Cobbles	Sand	Silt/mud	1	1	Poached	1	0	0	Improved grassland
24/07/2006	Kempla House	Trout beck	369470	524453	198	23	3	0	3	1	0	0	Pebbles	Cobbles	Sand	1	0	None	1	0	20	Improved grassland
24/07/2006	Birk sike	Trout beck	361748	527654	109	18	0	1	1	0	0	0	Silt/mud	Gravel		1	2	None	1	5	0	Improved grassland
24/07/2006	Eller beck	Trout	369203	525486	191	18	12	0	12	3	0	2	Gravel	Sand	Pebbles	0	0	None	0	2	0	Improved grassland
24/07/2006	Eller beck	Trout	368841	525562	172	18	37	1	38	0	1	8	Cobbles	Pebbles		0	2	Poached	1	0	65	Improved grassland
24/07/2006	Keisley Beck	Trout	371754	523435	172	16	17	73	90	1	1	15	Cobbles	Boulders	Pebbles	1	3	None	0	1	75	Improved grassland
14/08/2006	Mires beck	Glenridding	337877	516815	160	15	3	0	3	1	0	1	Boulders	Cobbles	Pebbles	0	1	None	0	0	70	Improved grassland
14/08/2006	Unamed beckne Grisedale	Grisedale	338615	518911	160	17	72	0	72	5	0	10	Pebbles	Gravel		0	0	Undercut	0	0	10	Unimproved grassland
15/08/2006	Unamed beck off Belah	Belah	388560	508910	423	14	0	0	0	0	0	0	Silt/mud	Gravel	Cobbles	1	1	Undercut	1	0	0	Unimproved grassland
15/08/2006	Mousegill Beck	Belah	385456	511750	424	14	0	0	0	0	0	0	Gravel	Cobbles	Silt/mud	1	1	Undercut	1	0	0	Unimproved grassland
15/08/2006	Argill	Belah	385232	513594	424	12	9	0	9	2	0	3	Cobbles	Pebbles	Boulders	1	2	Undercut	1	0	70	Improved grassland
16/08/2006	Gill beck	Glassonby	361064	538882	200	16	11	0	11	1	0	4	Cobbles	Pebbles		1	1	Undercut	1	0	70	Unimproved grassland
16/08/2006	Melmerby beck	Robbery	361698	537265	199	15	31	0	31	1	0	3	Cobbles	Boulders	Bedrock	0	2	Earth cliff	0	0	60	Urban/industrial
16/08/2006	Eller beck	Glassonby	361412	541302	232	13	8	0	8	1	0	4	Cobbles	Boulders	Pebbles	1	1	None	1	0	60	Improved grassland
16/08/2006	Gamblesby	Glassonby	361124	539621	233	16	24	0	24	3	0	4	Gravel	Pebbles	Silt/mud	1	0	Poached	1	0	0	Improved grassland
16/08/2006	Gamblesby	Glassonby	360989	539550	215	16	15	0	15	2	0	2	Pebbles	Cobbles	Gravel	0	0	None	0	0	0	Park/garden
16/08/2006	Limekiln Beck	Glassonby	362573	539831	424	10	2	0	2	3	0	1	Boulders	Bedrock	Cobbles	0	1	Undercut	1	0	5	Unimproved grassland
16/08/2006	Limekiln/Hazelrigg	Glassonby	361362	540327	424	11	20	0	20	1	0	10	Cobbles	Pebbles	Gravel	1	2	Poached	1	0	30	Improved grassland
16/08/2006	Eller beck	Glassonby	360466	540679	176	13	11	0	11	3	0	2	Cobbles	Boulders	Pebbles	0	1	Undercut	0	0	50	Arable crop
17/08/2006	Scarrowmanwick beck	Croglin	358202	547264	224	14	0	0	0	0	0	0	Gravel	Pebbles	Cobbles	1	1	Undercut	1	0	10	Unimproved grassland
17/08/2006	Croglin main river	Croglin	358147	547349	229	13	2	0	2	3	0	1	Boulders	Cobbles	Pebbles	0	4	Earth cliff	1	0	50	Unimproved grassland
17/08/2006	Small Croglin beck	Croglin	357629	547341	207	16	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	1	1	Poached	1	0	0	Improved grassland
18/08/2006	Lino Gill	Croglin	357577	546735	207	16	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	1	1	Poached	1	0	0	Improved grassland
18/08/2006	Ainstable beck	Eden Lower	352583	546542	111	14	5	0	5	2	0	1	Cobbles	Pebbles	Gravel	1	2	Poached	1	0	80	Improved grassland
18/08/2006	Little Briggle beck	Croglin	355605	544985	129	15	6	0	6	1	0	2	Cobbles	Pebbles	Gravel	1	1	None	1	0	0	Improved grassland
18/08/2006	Briggle beck	Croglin	357133	545594	168	15	2	0	2	2	0	1	Sand	Pebbles	Gravel	1	0	Poached	1	0	0	Unimproved grassland
18/08/2006	Force beck	Lowther	357620	513814	171	14	0	0	0	3	0	0	Boulders	Cobbles	Gravel	1	2	Poached	1	0	70	Improved grassland
18/08/2006	Force beck d/s waterfall	Lowther	358753	513835	171	13	17	0	17	2	0	2	Cobbles	Pebbles	Gravel	0	2	None	0	0	80	Woodland
18/08/2006	Force beck u/s motorway	Lowther	358004	513325	171	13	0	0	0	0	0	0	Bedrock	Boulders	Cobbles	0	2	Undercut	1	0	0	Unimproved grassland
24/07/2006	Murton beck	Trout	372606	521680	242	22	81	0	81	1	0	20	Gravel	Pebbles	Cobbles	0	3	None	0	0	40	Urban/industrial
24/07/2006	Dufton Ghyll	Trout	369031	524850	216	17	37	0	37	3	0	12	Cobbles	Boulders	Gravel	0	2	Earth cliff	0	0	95	Woodland
24/07/2006	Dufton Ghyll	Trout	369448	524551	196	23	19	0	19	0	0	8	Cobbles	Pebbles	Gravel	1	1	Poached	0	0	0	Improved grassland
25/07/2006	Trout	Trout	365313	524379	109	18	0	17	17	2	1	6	Pebbles	Cobbles	Gravel	0	7	None	0	5	30	Improved grassland
25/07/2006	Trout	Trout	366610	524340	0	19	0	55	55	2	5	25	Cobbles	Boulders	Pebbles	0	6	None	0	5	30	Improved grassland
25/07/2006	Murton beck	Trout	370914	521384	109	22	32	0	32	6	0	12	Cobbles	Boulders	Pebbles	1	1	Poached	1	0	5	Improved grassland
26/07/2006	Trout	Trou	363945	525463	108	18	1	24	25	0	1	8	Boulders	Cobbles	Gravel	0	7	None	0	5	30	Improved grassland
26/07/2006	Keld sike	Trout	366701	523760	129	21	0	0	0	0	0	0	Sand	Cobbles	Pebbles	1	1	None	0	2	30	Improved grassland
26/07/2006	Keld sike	Trout	366505	524032	131	20	0	0	0	0	0	0	Sand	Cobbles		1	1	None	0	1	20	Improved grassland

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							No of fish caugh in 5 minutes of fishing															
Date of survey	Site	Sub-catchment	Easting	Northing	Altitude (m above sea level)	Water temp °C	0+ Trout	0+ Salmon	Total 0+ salmon and trout	1+ and older Trout	1+ and older Salmon	No. of 0+ fish missed	Dominant substrate	Substrate 2	Substrate 3	Siltation 1=present 0=absent	River width (m)	Erosion type	Stock access 1=yes 0=no	Fence width (m)	% Channel Cover	Landuse
26/07/2006	Trout	Trout	368594	523410	136	21	6	12	18	0	0	5	Cobbles	Pebbles	Gravel	1	2	Earth cliff	0	0	90	Improved grassland
26/07/2006	Keysley Beck	Trout	370273	522902	136	20	12	9	21	1	0	6	Boulders	Cobbles	Gravel	0	3	None	0	0	70	Woodland
26/07/2006	Trout	Trout	368565	525303	158	17	15	10	25	0	1	6	Cobbles	Pebbles	Gravel	0	3	None	0	0	90	Woodland
26/07/2006	Stank beck	Crowdundle	365466	528968	158	19	38	0	38	1	1	5	Cobbles	Pebbles	Gravel	1	2	None	0	0	90	Improved grassland
27/07/2006	Trout	Trout	368394	526952	158	16	44	1	45	2	0	8	Boulders	Cobbles	Pebbles	0	3	Poached	1	2	75	Improved grassland
27/07/2006	Murton beck	Trout	370238	522027	158	17	17	0	17	0	0	3	Pebbles	Cobbles	Gravel	1	3	Undercut	0	20	90	Unimproved grassland
27/07/2006	Milburn	Crowdundle	367105	528156	158	20	35	1	36	0	1	10	Cobbles	Pebbles	Gravel	1	5	None	1	0	70	Unimproved grassland
27/07/2006	Milburn	Crowdundle	365953	528628	196	19	8	18	26	0	0	10	Cobbles	Pebbles	Gravel	1	4	Undercut	0	0	70	Unimproved grassland
27/07/2006	Stank beck	Crowdundle	364757	529138	198	18	21	0	21	1	0	7	Sand	Pebbles	Cobbles	1	3	Poached	1	0	80	Unimproved grassland
27/07/2006	Crowdundle	Crowdundle	360984	528078	196	20	0	39	39	0	0	7	Cobbles	Pebbles	Gravel	0	5	None	0	0	70	Improved grassland
28/07/2006	Blind beck	Lyvennet	361140	518547	180	20	0	0	0	0	0	0	Gravel	Silt/mud		1	1	Undercut	1	0	0	Improved grassland
28/07/2006	Crooks Beck	Scandal	374410	502988	288	16	0	0	0	0	0	0	Cobbles	Pebbles		1	0	None	1	0	0	Unimproved grassland
28/07/2006	Artlegarth	Scandal	372156	503563	266	21	1	0	1	1	0	0	Cobbles	Bedrock	Pebbles	0	2	None	0	0	0	Unimproved grassland
28/07/2006	Thackthwaite	Scandal	372169	503068	266	18	0	0	0	1	0	0	Cobbles	Boulders	Bedrock	0	1	None	1	0	0	Unimproved grassland
28/07/2006	Wyegarth	Scandal	372019	503556	270	20	5	0	5	2	0	2	Cobbles	Pebbles	Gravel	0	1	None	1	0	0	Unimproved grassland
28/07/2006	Scandal	Scandal	374261	509786	271	16	1	7	8	1	0	3	Cobbles	Boulders	Pebbles	0	5	None	0	6	90	Improved grassland
28/07/2006	Low wodd gill	Lyvennet	362168	517009	168	20	2	3	5	1	0	0	Cobbles	Boulders	Sand	1	2	Poached	1	0	0	Improved grassland
31/07/2006	Upper Belah	Belah	384757	509429	285	16	6	0	6	0	0	1	Bedrock	Cobbles	Gravel	0	2	Earth cliff	1	0	0	Improved grassland
31/07/2006	Scandal at Soulby Mill	Scandal	374633	510189	168	16	1	18	19	1	0	6	Cobbles	Boulders	Pebbles	0	5	Poached	1	0	0	Improved grassland
31/07/2006	Scandal at Smardale	Scandal	373573	508636	166	15	2	34	36	0	1	15	Cobbles	Boulders	Pebbles	0	10	None	0	3	50	Improved grassland
31/07/2006	Coldkeld Sike	Belah	382617	510338	266	13	12	0	12	9	0	6	Cobbles	Pebbles	Boulders	1	2	None	1	0	80	Improved grassland
01/08/2006	Hole Sike	Briggle	361743	534459	150	0	0	0	0	0	0	0	Silt/mud	Cobbles	Gravel	1	1	None	0	0	10	Park/garden
01/08/2006	Upper Crowdundle	Crowdundle	368875	531261	193	12	4	6	10	0	0	4	Boulders	Cobbles	Pebbles	0	4	Undercut	1	0	0	Unimproved grassland
01/08/2006	Middle Tongue beck	Crowdundle	367506	531279	298	13	37	0	37	0	0	6	Gravel	Cobbles	Boulders	0	1	Undercut	1	0	30	Unimproved grassland
01/08/2006	Aigill Sike	Belah	360075	532435	117	14	10	0	10	1	0	3	Gravel	Pebbles	Cobbles	1	1	None	0	10	90	Improved grassland
01/08/2006	Crowdundle	Crowdundle	360736	528253	107	14	0	63	63	0	1	6	Cobbles	Pebbles	Gravel	0	7	None	0	5	70	Improved grassland
01/08/2006	Blencarn beck	Briggle	364272	531386	107	0	0	0	0	0	0	0	Sand	Gravel	Cobbles	1	1	None	1	0	0	Improved grassland
01/08/2006	Ran Beck	Briggle	364392	531467	189	14	4	0	4	1	0	1	Cobbles	Gravel		1	1	Poached	1	0	80	Improved grassland
03/08/2006	Keld Gill	Lowther	354516	514004	245	17	7	0	7	0	2	2	Boulders	Cobbles	Pebbles	0	2	None	1	0	0	Unimproved grassland
03/08/2006	Lowther	Lowther	351674	518868	245	17	0	51	51	0	0	10	Cobbles	Pebbles	Gravel	0	10	None	0	5	5	Improved grassland
03/08/2006	Hobgrumble	Lowther (swindale)	350251	511821	263	19	9	1	10	0	0	4	Cobbles	Pebbles	Gravel	0	2	Undercut	1	0	0	Unimproved grassland
03/08/2006	Force Gill	Lowther	356399	514226	258	13	29	1	30	0	0	8	Cobbles	Gravel	Sand	1	2	Undercut	1	0	0	Improved grassland
03/08/2006	Unamed Lowther trib	Lowther	355657	513150	254	14	0	0	0	0	0	0	Boulders	Cobbles		0	1	Poached	1	0	0	Unimproved grassland
03/08/2006	Thornship Gill	Lowther	355280	513479	246	16	2	0	2	0	2	0	Boulders	Cobbles	Pebbles	0	1	Undercut	1	0	0	Unimproved grassland
03/08/2006	Mosedale beck	Lowther	350406	511794	266	19	2	0	2	0	0	0	Boulders	Cobbles	Pebbles	0	5	Undercut	1	0	5	Unimproved grassland
03/08/2006	Gouthercrag Gill	Lowther	351409	513025	269	16	1	0	1	0	0	0	Cobbles	Boulders	Pebbles	0	3	None	1	1	80	Improved grassland
25/08/2006	Naddle beck	Lowther	351026	515931	205	14	0	0	0	0	0	0	Boulders	Cobbles	Gravel	0	1	None	0	2	90	Improved grassland
25/08/2006	Littlewater beck	Lowther	351597	517719	211	12	0	0	0	1	0	0	Cobbles	Boulders	Pebbles	1	2	Poached	1	0	80	Improved grassland
25/08/2006	Cawdale beck	Howe	349669	517796	258	16	8	0	8	1	0	2	Boulders	Bedrock	Cobbles	0	2	Undercut	1	0	0	Unimproved grassland
25/08/2006	Willdale beck	Howe	349685	517701	279	15	3	0	3	1	0	2	Boulders	Bedrock	Cobbles	0	1	Undercut	1	0	0	Unimproved grassland
25/08/2006	Cawdale	Howes	349210	518046	323	17	5	0	5	1	0	3	Boulders	Cobbles	Gravel	0	2	Undercut	1	0	0	Unimproved grassland
29/08/2006	Lightwater beck	Eamont	354796	529130	159	14	0	0	0	0	0	0	Pebbles	Gravel	Sand	1	1	Undercut	0	10	70	Improved grassland
29/08/2006	Lady beck	Eamont	350140	527025	159	13	0	0	0	4	0	0	Cobbles	Boulders	Pebbles	1	1	None	1	0	75	Improved grassland
29/08/2006	Whale beck	Lowther	351801	521440	177	14	0	0	0	0	0	1	Cobbles	Boulders	Pebbles	0	2	Poached	1	0	50	Improved grassland
30/08/2006	Melmerby beck	Robbery	362690	537100	220	12	47	0	47	2	0	7	Gravel	Pebbles	Cobbles	0	2	Undercut	0	0	90	Conifer plantation
30/08/2006	Seavy sike	Raven	362300	542664	317	11	5	0	5	1	0	1	Cobbles	Boulders	Pebbles	1	0	Undercut	1	0	0	Unimproved grassland
30/08/2006	Swarthgill beck	Raven	360888	542616	261	12	0	0	0	0	0	0	Pebbles	Gravel	Sand	1	0	Undercut	1	0	0	Improved grassland
30/08/2006	Mill beck	Eden Lower	350240	545414	67	13	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	1	1	None	0	0	20	Park/garden
18/08/2006	Heltondale	Heltondale	349242	520181	279	12	17	0	17	0	0	6	Cobbles	Pebbles	Gravel	0	3	None	0	5	80	Improved grassland
18/08/2006	Setterah Syke	Heltondale	350708	520370	276	14	4	0	4	0	1	0	Cobbles	Pebbles	Gravel	1	1	Undercut	1	0	50	Unimproved grassland
22/08/2006	Coegin beck	Aira	337112	520700	392	12	7	0	7	3	0	2	Boulders	Cobbles	Pebbles	0	1	Undercut	1	0	0	Improved grassland
22/08/2006	Upper Aira	Aira	336867	520859	396	12	2	0	2	0	0	0	Boulders	Cobbles	Pebbles	0	8	Undercut	1	0	0	Unimproved grassland
22/08/2006	Ridding beck	Aira	339722	521716	277	14	0	0	0	1	0	0	Cobbles	Boulders	Pebbles	1	1	Undercut	0	0	0	Unimproved grassland
22/08/2006	Southwaite	Dacre	344281	525572	241	15	0	0	0	0	0	0	Bedrock	Cobbles	Pebbles	1	3	Poached	1	0	80	Improved grassland
22/08/2006	Swinescale	Dacre	341711	527490	250	15	24	0	24	16	0	6	Cobbles	Boulders	Pebbles	0	1	Undercut	1	0	0	Improved grassland
22/08/2006	Cooper beck	Dacre	340440	527325	271	16	3	48	51	0	1	10	Cobbles	Pebbles		1	2	Undercut	1	0	0	Improved grassland

2006 Survey: Semi-quantitative electrofishing with habitat data (Eden Rivers Trust)

							No of fish caugh in 5 minutes of fishing															
Date of survey	Site	Sub-catchment	Easting	Northing	Attitude (m above sea level)	Water temp °C	0+ Trout	0+ Salmon	Total 0+ salmon and trout	1+ and older Trout	1+ and older Salmon	No. of 0+ fish missed	Dominant substrate	Substrate 2	Substrate 3	Siltation 1=present 0=absent	River width (m)	Erosion type	Stock access 1=yes 0=no	Fence width (m)	% Channel Cover	Landuse
22/08/2006	Lowthwaite beck	Dacre	340892	523560	269	16	9	2	11	2	0	4	Cobbles	Pebbles	Gravel	1	1	Undercut	1	0	0	Improved grassland
22/08/2006	Greaves beck	Dacre	343741	525079	241	15	5	0	5	0	0	2	Cobbles	Pebbles	Gravel	1	1	Poached	1	0	50	Improved grassland
21/08/2006	Cambeck u/s pipebridge	Cambeck	353478	568926	190	14	3	0	3	0	0	1	Cobbles	Pebbles		0	4	None	0	0	90	Woodland
15/08/2006	Powbrand sike	Belah	381288	513338	424	14	19	0	19	3	0	8	Gravel	Pebbles		1	1	Undercut	0	2	0	Improved grassland
15/08/2006	Augil	Belah	381673	514860	424	13	7	0	7	3	0	3	Cobbles	Boulders	Pebbles	1	2	None	0	4	80	Woodland
15/08/2006	Upper Belah	Belah	386599	508766	423	13	2	0	2	4	0	0	Gravel	Cobbles	Pebbles	1	1	Undercut	1	0	0	Unimproved grassland
15/08/2006	Upper Belah	Belah	386012	508727	422	0	0	0	0	0	0	0	Bedrock	Cobbles	Gravel	1	1	Undercut	1	0	0	Unimproved grassland
15/08/2006	Bleabury beck	Belah	386019	508684	415	14	2	0	2	5	0	1	Boulders	Cobbles	Gravel	1	1	Undercut	1	0	0	Unimproved grassland
14/08/2006	Upper Kirkstone	Kirkstone	340099	510684	158	12	19	0	19	0	3	5	Cobbles	Boulders	Pebbles	0	1	Undercut	1	0	0	Unimproved grassland
14/08/2006	Angletarn Beck	Goldril	340588	514090	158	14	46	0	46	3	0	6	Cobbles	Pebbles	Gravel	0	1	None	1	0	0	Improved grassland
14/08/2006	Stonebarrow Ghyll	Goldril	340158	515626	158	14	14	0	14	0	0	4	Cobbles	Pebbles	Gravel	0	1	Earth cliff	1	0	30	Improved grassland
14/08/2006	Groovegill	Ullswater	339527	520019	182	17	23	0	23	3	0	8	Cobbles	Pebbles	Gravel	1	1	Undercut	1	0	10	Unimproved grassland
14/08/2006	Rampsgill U/S road	Ullswater	345454	523604	163	16	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	1	1	Undercut	0	0	80	Park/garden
14/08/2006	Rampsgill d/s road barrier	Ullswater	345569	523517	163	16	22	0	22	1	0	6	Cobbles	Boulders	Pebbles	0	1	Undercut	0	5	90	Improved grassland
14/08/2006	Hag beck	Grisedale	338983	515904	159	14	33	0	33	2	1	6	Pebbles	Cobbles	Gravel	0	1	Undercut	0	5	80	Unimproved grassland
04/08/2006	Castle carrock beck	Gelt	353962	558166	121	16	24	0	24	4	0	6	Pebbles	Cobbles	Gravel	0	2	Poached	1	1	40	Improved grassland
04/08/2006	Carrdoaning	Cairn	354055	550416	121	14	0	0	0	3	0	0	Gravel	Cobbles	Boulders	0	3	None	0	0	90	Woodland
04/08/2006	Milton at Cumcatch	Milton	354775	561091	120	14	20	0	20	0	0	5	Pebbles	Cobbles	Gravel	0	3	Poached	0	5	0	Improved grassland
04/08/2006	Boggle beck	Gelt	355129	557624	121	15	2	0	2	0	0	1	Pebbles	Cobbles	Gravel	1	2	Earth cliff	1	0	90	Woodland
04/08/2006	Mill beck	Gelt	355427	556447	121	14	22	0	22	5	0	5	Cobbles	Boulders	Pebbles	0	2	Poached	1	0	70	Improved grassland
07/08/2006	Powdonnet Well	Lyvennet	359727	522852	140	12	0	0	0	0	0	0	Cobbles	Boulders	Pebbles	1	2	Poached	1	0	0	Improved grassland
07/08/2006	Leith at Rectory farm	Leith	358857	524361	120	19	1	10	11	0	0	1	Cobbles	Pebbles		0	4	None	0	5	0	Improved grassland
07/08/2006	Leith at Brown howe	Leith	356626	525224	121	16	2	0	2	0	0	1	Cobbles	Boulders	Sand	1	5	None	0	3	0	Improved grassland
07/08/2006	Leith at Woodhouse	Leith	357169	524943	126	18	0	8	8	0	1	0	Cobbles	Boulders		1	3	Poached	1	0	0	Improved grassland
07/08/2006	Upper Morland	Lyvennet	359758	522856	138	17	1	0	1	0	0	0	Cobbles	Boulders	Gravel	1	3	Poached	1	0	10	Improved grassland
08/08/2006	Leith at boulders at quarry	Leith	355501	517722	212	18	2	0	2	3	0	1	Silt/mud	Gravel	Boulders	1	2	None	0	0	0	Urban/industrial
08/08/2006	Leith at high hall	Leith	356080	519071	212	16	10	0	10	7	0	3	Cobbles	Boulders	Pebbles	1	3	None	0	0	0	Improved grassland
08/08/2006	Leith at Little Strickland	Leith	355797	520696	212	12	10	0	10	2	0	2	Cobbles	Boulders	Gravel	1	5	Poached	1	0	0	Improved grassland
08/08/2006	Leith at Sayers	Leith	355050	523900	0	0	9	0	9	1	0	2	Boulders	Cobbles	Pebbles	1	3	Earth cliff	0	0	0	Improved grassland
08/08/2006	Leith at quarry ranunc plant s	Leith	355602	518331	212	14	10	0	10	0	0	2	Cobbles	Pebbles	Gravel	1	2	None	0	0	0	Woodland
09/08/2006	Dalebank	Lyvennet	362144	514840	145	14	2	0	2	1	0	0	Pebbles	Cobbles	Gravel	0	1	None	0	0	0	Park/garden
09/08/2006	Leith at Leith House	Leith	358199	524574	113	15	0	21	21	0	0	6	Bedrock	Cobbles	Pebbles	1	3	None	0	5	50	Improved grassland
09/08/2006	Lyvennet at Skygarth	Lyvennet	360850	525899	95	15	0	36	36	0	0	10	Cobbles	Pebbles	Gravel	0	7	None	0	10	0	Improved grassland
09/08/2006	Lyvennet at Crossrigg	Lyvennet	361185	523387	123	15	0	19	19	1	0	6	Bedrock	Boulders	Cobbles	0	4	None	0	5	0	Improved grassland
09/08/2006	Lyvennet at Kempley	Lyvennet	361444	522347	126	15	0	9	9	0	0	4	Cobbles	Boulders	Pebbles	0	4	None	1	3	20	Improved grassland
09/08/2006	Lyvennet at High Whitber	Lyvennet	361980	520138	127	17	0	23	23	0	0	6	Cobbles	Boulders	Pebbles	0	7	None	0	5	40	Improved grassland
09/08/2006	Little beck	Lyvennet	361977	520190	138	17	0	0	0	0	0	0	Silt/mud	Cobbles	Gravel	0	1	Undercut	0	2	0	Improved grassland
10/08/2006	Pencilmillock Beck	Ullswater	343364	522300	221	15	54	0	54	2	0	15	Boulders	Cobbles	Gravel	1	2	Undercut	0	0	90	Improved grassland
21/08/2006	Cambeck d/s pipebridge	Cambeck	352652	567546	113	15	15	0	15	0	0	5	Cobbles	Boulders	Pebbles	0	6	None	0	0	75	Unimproved grassland
21/08/2006	Cambeck d/s weir	Cambeck	350955	562573	31	15	2	20	22	0	1	6	Cobbles	Pebbles	Gravel	0	6	None	0	0	70	Improved grassland
21/08/2006	Peglands beck	Kingwater	352760	565238	31	15	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	1	1	Poached	1	5	80	Improved grassland
21/08/2006	Mill beck	Kingwater	358291	566966	31	17	4	0	4	9	0	2	Cobbles	Pebbles	Gravel	1	1	Undercut	0	2	50	Park/garden
23/08/2006	Rockcliffe beck	Eden Lower	335956	561656	18	16	0	0	0	0	0	0	Cobbles	Pebbles		1	2	None	0	2	30	Park/garden
23/08/2006	Cargo beck	Eden Lower	337349	559290	17	16	0	0	0	0	0	0	Cobbles	Pebbles		1	2	None	0	0	60	Arable crop
23/08/2006	Brunstock beck	Eden lower	341526	559861	29	15	0	0	0	0	0	0	Silt/mud	Sand		1	3	None	0	5	5	Improved grassland
23/08/2006	Troutbeck	Cairn	349073	555855	31	15	0	0	0	0	0	0	Sand	Pebbles	Cobbles	1	1	None	0	2	50	Improved grassland
23/08/2006	Hawkney beck	Cairn	349477	556815	56	14	0	0	0	0	0	0	Pebbles	Cobbles	Gravel	1	1	Earth cliff	1	0	0	Improved grassland
23/08/2006	Chapelwell beck	Eden Lower	351427	550482	56	16	0	0	0	0	0	0	Silt/mud	Boulders	Cobbles	1	1	None	0	0	95	Woodland
23/08/2006	Brunstock beck	Eden Lower	341150	557001	29	16	0	0	0	0	2	0	Silt/mud	Bedrock		1	3	None	0	0	50	Park/garden
23/08/2006	Milbeck	Eden Lower	336056	557927	0	0	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	1	1	None	0	0	80	Improved grassland
23/08/2006	Corby beck	Eden Lower	347070	554957	33	15	0	0	0	0	0	0	Cobbles	Boulders	Silt/mud	1	1	None	0	10	90	Woodland
24/08/2006	Small beck off mardale	Haweswater	346791	510808	223	18	38	0	38	3	0	4	Pebbles	Cobbles	Gravel	0	0	Undercut	0	20	0	Unimproved grassland
24/08/2006	Gatesgarth beck	Haweswater	346918	510783	235	17	21	0	21	3	0	5	Boulders	Cobbles	Pebbles	0	3	Undercut	0	20	0	Unimproved grassland
24/08/2006	Unamed Haweswater beck	Haweswater	346882	511694	261	16	10	0	10	3	0	3	Pebbles	Cobbles	Gravel	0	1	Undercut	1	0	0	Unimproved grassland
24/08/2006	Small beck off Mardale beck	Haweswater	346623	510735	255	17	21	0	21	4	0	3	Pebbles	Cobbles	Gravel	0	1	Undercut	1	0	0	Unimproved grassland
24/08/2006	Mardale beck	Haweswater	346651	510756	254	18	3	0	3	0	0	1	Boulders	Cobbles		0	3	Undercut	1	0	0	Unimproved grassland

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							No of fish caugh in 5 minutes of fishing																
Date of survey	Site	Sub-catchment	Easting	Northing	Altitude (m above sea level)	Water temp °C	0+ Trout	0+ Salmon	Total 0+ salmon and trout	1+ and older Trout	1+ and older Salmon	No. of 0+ fish missed	Dominant substrate	Substrate 2	Substrate 3	Siltation 1=present 0=absent	River width (m)	Erosion type	Stock access 1=yes 0=no	Fence width (m)	% Channel Cover	Landuse
24/08/2006	Randale beck	Haweswater	346868	511843	247	13	4	0	4	0	0	0	Boulders	Cobbles		0	3	Undercut	0	10	10	Unimproved grassland
24/08/2006	Riggindale beck	Haweswater	346331	511782	274	15	9	0	9	2	0	3	Boulders	Cobbles	Gravel	0	3	Undercut	1	0	0	Unimproved grassland
24/08/2006	Riggindale beck	Haweswater	346786	511778	262	16	3	0	3	0	0	1	Boulders	Cobbles		0	3	Undercut	1	0	0	Unimproved grassland
25/08/2006	Little Mossy beck	Heltondale	349025	521135	303	18	4	0	4	0	0	1	Cobbles	Boulders	Pebbles	0	1	Undercut	1	0	0	Unimproved grassland
25/08/2006	Naddle beck	Lowther	350417	514350	336	10	3	0	3	3	0	1	Cobbles	Boulders		0	1	Undercut	1	0	0	Unimproved grassland
01/09/2006	Hawkney beck	Cairn	351009	556547	74	13	8	0	0	1	0	0	Cobbles	Pebbles	Gravel	1	1	None	0	0	0	Park/garden
01/09/2006	Hawkney	Cairn	350899	556397	61	14	4	0	4	0	0	1	Cobbles	Pebbles	Gravel	0	1	None	0	0	0	Park/garden
01/09/2006	Pottlebog	Irthing	350213	558872	61	14	0	0	0	0	0	0	Sand			1	1	Undercut	1	0	80	Improved grassland
01/09/2006	Trout beck	Cairn	350836	555339	92	14	0	0	0	0	0	0	Sand	Gravel	Cobbles	1	1	Undercut	0	10	90	Woodland
01/09/2006	Lime Kiln beck	Gelt	358766	554719	93	10	0	0	0	0	0	0	Boulders	Gravel		1	1	Undercut	1	0	0	Unimproved grassland
01/09/2006	Hell beck	Gelt	353850	557825	93	14	27	0	27	2	0	8	Pebbles	Cobbles	Gravel	1	2	Undercut	1	0	95	Woodland
01/09/2006	Willy beck	Gelt	354743	557697	103	14	8	0	8	0	0	2	Cobbles	Pebbles	Gravel	1	1	Undercut	0	0	80	Woodland
01/09/2006	Cumrew Beck	Cairn	356065	550730	181	15	6	0	6	1	0	2	Pebbles	Cobbles	Gravel	1	1	Undercut	0	3	90	Improved grassland
04/09/2006	Carrock beck	Caldew	333130	534906	408	12	9	0	9	0	0	2	Gravel	Pebbles	Cobbles	0	1	Undercut	1	0	0	Unimproved grassland
04/09/2006	Gillcambdon beck	Caldew	337546	536795	256	14	0	0	0	0	0	0	Boulders	Cobbles	Bedrock	0	3	Earth cliff	0	0	80	Woodland
04/09/2006	Gillcambdon beck	Caldew	338437	537473	197	15	1	2	3	0	0	0	Cobbles	Boulders	Pebbles	0	4	None	0	2	80	Improved grassland
04/09/2006	Carrock beck	Caldew	335014	535024	238	14	7	9	16	0	2	2	Boulders	Cobbles	Pebbles	0	3	Undercut	1	0	0	Unimproved grassland
04/09/2006	Heggle syke	Caldew	338687	536048	198	12	3	12	15	5	0	5	Cobbles	Pebbles	Gravel	0	2	Poached	1	0	40	Improved grassland
04/09/2006	Heggle syke	Caldew	338867	534788	198	14	0	0	0	0	0	0	Gravel	Pebbles	Cobbles	1	1	Undercut	0	0	95	Woodland
04/09/2006	Gillcambdon beck	Caldew	338710	534936	259	15	0	0	0	0	0	0	Cobbles	Boulders		1	2	Poached	1	0	5	Improved grassland
05/09/2006	Gillfoot beck	Caldew	334820	538424	250	17	0	0	0	1	1	0	Cobbles	Boulders	Pebbles	1	1	Poached	1	0	0	Improved grassland
05/09/2006	Stott Ghyll	Caldew	334572	538543	250	15	0	0	0	0	0	0	Boulders	Cobbles	Pebbles	0	2	Earth cliff	1	1	25	Improved grassland
05/09/2006	Busta beck	Caldew	337541	541728	127	14	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	1	1	None	1	0	50	Woodland
05/09/2006	Banty beck	Eden Lower	354928	535946	0	14	0	0	0	0	0	0	Boulders	Cobbles	Sand	1	1	Poached	0	0	5	Improved grassland
05/09/2006	Whitewell	Caldew	334174	537029	262	14	1	0	1	0	0	0	Pebbles	Cobbles	Gravel	1	0	Poached	1	0	0	Improved grassland
05/09/2006	Blea gill	Caldew	334309	537136	249	15	6	0	6	2	0	2	Cobbles	Boulders	Pebbles	1	1	Poached	1	0	0	Improved grassland
05/09/2006	Hud Beck	Caldew	337206	543850	120	15	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	1	2	None	1	0	90	Arable crop
05/09/2006	Powbeck	Caldew	339275	548719	102	16	0	0	0	0	0	0	Boulders	Cobbles	Gravel	1	2	Poached	1	0	75	Improved grassland
06/09/2006	Ive at High Head	Ive	340841	543704	100	16	1	0	1	0	0	0	Cobbles	Pebbles	Silt/mud	1	3	None	0	0	5	Park/garden
06/09/2006	Ive at the Grange	Ive	341753	542995	100	16	0	0	0	1	0	0	Cobbles	Pebbles	Gravel	0	4	None	0	0	50	Park/garden
06/09/2006	Grilse beck	Ive	342553	540054	100	17	0	0	0	0	0	0	Cobbles	Gravel	Silt/mud	1	1	Poached	1	0	50	Improved grassland
06/09/2006	Line beck	Ive	343015	541994	145	16	0	0	0	0	0	0	Cobbles	Boulders		1	2	None	0	0	95	Arable crop
07/09/2006	Roe	Roe	338778	545078	95	15	3	2	5	0	1	1	Bedrock	Cobbles	Pebbles	0	5	Earth cliff	0	0	70	Woodland
07/09/2006	Powburgh beck	Eden lower	339815	548020	115	15	0	0	0	0	0	0	Cobbles	Pebbles	Sand	1	1	Earth cliff	0	0	90	Woodland
07/09/2006	Cuthwaite	Roe	339682	543219	109	15	0	0	0	0	0	0	Cobbles	Pebbles	Sand	1	1	Undercut	0	10	40	Unimproved grassland
07/09/2006	Roe	Roe	339579	543360	96	16	0	0	0	0	0	0	Cobbles	Pebbles	Sand	1	3	Earth cliff	0	0	70	Improved grassland
07/09/2006	Fellican beck	Roe	338960	543693	95	14	0	0	0	0	0	0	Cobbles	Pebbles	Sand	1	1	Poached	1	0	70	Improved grassland
07/09/2006	Roe	Roe	341073	536790	149	13	0	0	0	0	0	0	Cobbles	Boulders	Silt/mud	1	2	Earth cliff	0	0	80	Improved grassland
07/09/2006	Roe	Roe	339358	541723	114	14	2	0	2	0	0	0	Cobbles	Bedrock	Pebbles	1	3	Poached	1	0	75	Improved grassland
07/09/2006	Rush gill	Roe	340410	538500	150	13	0	0	0	0	0	0	Bedrock	Cobbles	Gravel	1	1	None	0	0	90	Woodland
08/09/2006	Eastfield sike	Hayber	376295	518559	183	11	4	0	4	0	0	2	Cobbles	Pebbles	Gravel	0	1	Undercut	1	0	0	Unimproved grassland
08/09/2006	Deep Gill	Hayber	375644	516805	173	11	12	0	12	8	0	4	Cobbles	Pebbles	Gravel	0	2	Undercut	1	0	0	Unimproved grassland
08/09/2006	Hayber above barrier	Hayber	375216	516567	173	12	9	0	9	4	0	0	Cobbles	Pebbles	Gravel	0	3	Undercut	1	0	0	Unimproved grassland
08/09/2006	Hayber gill beloew barrier	Hayber	374953	516341	165	15	4	0	4	5	0	0	Cobbles	Pebbles	Gravel	1	2	Undercut	0	0	0	Park/garden
08/09/2006	Hayber below A66	Hayber	374804	515450	164	14	2	20	22	0	0	4	Cobbles	Pebbles	Gravel	0	3	None	0	0	50	Park/garden
08/09/2006	Gaisgill	Hilton	371197	518934	142	15	11	17	28	8	0	6	Cobbles	Pebbles	Sand	1	2	Undercut	1	0	0	Improved grassland
11/09/2006	Gill beck	Caldbeck	332069	538902	195	17	5	0	5	7	0	2	Cobbles	Boulders	Pebbles	0	3	None	1	0	0	Improved grassland
11/09/2006	Gill beck	Caldbeck	331856	537397	274	13	25	0	25	3	0	4	Pebbles	Cobbles	Gravel	0	2	None	0	10	80	Woodland
11/09/2006	Gill beck	Caldbeck	332071	537758	264	16	1	0	1	1	0	0	Cobbles	Boulders	Pebbles	0	2	Undercut	1	0	60	Unimproved grassland
11/09/2006	Eller beck	Caldbeck	328844	538606	239	13	0	0	0	0	0	0	Cobbles	Pebbles		0	1	Undercut	1	0	0	Unimproved grassland
11/09/2006	Burblethwaite	Caldbeck	328986	537180	258	14	1	0	1	1	0	0	Cobbles	Pebbles	Gravel	0	3	Undercut	0	3	5	Improved grassland
11/09/2006	Burblethwaite	Caldbeck	327820	536368	313	13	2	0	2	6	0	0	Boulders	Cobbles	Pebbles	0	2	Undercut	1	0	0	Unimproved grassland
12/09/2006	Ravenbeck	Petteril	347403	539217	217	16	0	0	0	0	0	0	Silt/mud	Pebbles	Gravel	1	1	Poached	1	0	40	Improved grassland
12/09/2006	Petteril at Low ground	Petteril	349049	538799	219	17	0	0	0	1	0	0	Cobbles	Boulders	Pebbles	0	5	None	0	10	50	Improved grassland
12/09/2006	Gillcambdon	Caldew	336407	537461	262	14	8	5	13	0	1	4	Boulders	Cobbles	Pebbles	0	2	None	0	2	70	Improved grassland
12/09/2006	Caldew	Caldew	336330	537490	262	13	0	3	3	0	0	0	Boulders	Cobbles		0	8	None	0	2	5	Improved grassland

2006 Survey: Semi-quantitative electrofishing with habitat data (Eden Rivers Trust)

							No of fish caugh in 5 minutes of fishing															
Date of survey	Site	Sub-catchment	Easting	Northing	Altitude (m above sea level)	Water temp °C	0+ Trout	0+ Salmon	Total 0+ salmon and trout	1+ and older Trout	1+ and older Salmon	No. of 0+ fish missed	Dominant substrate	Substrate 2	Substrate 3	Siltation 1=present 0=absent	River width (m)	Erosion type	Stock access 1=yes 0=no	Fence width (m)	% Channel Cover	Landuse
12/09/2006	Greystoke beck	Petteril	343951	531081	0	16	8	0	8	1	0	2	Cobbles	Boulders	Pebbles	0	1	None	0	0	75	Park/garden
14/09/2006	Petteril at Petteril bank	Petteril	346322	542706	97	15	1	0	1	1	1	0	Cobbles	Pebbles		0	7	None	0	5	50	Improved grassland
14/09/2006	Woodside	Petteril	342916	550322	94	15	0	0	0	0	0	0	Cobbles	Pebbles	Silt/mud	1	1	Poached	1	0	30	Improved grassland
14/09/2006	Cowbrand	Petteril	341447	552829	56	15	0	0	0	0	0	0	Cobbles	Pebbles	Silt/mud	0	1	None	0	2	10	Improved grassland
14/09/2006	Brown beck	Heltondale	348576	519824	297	14	11	1	12	1	1	3	Cobbles	Boulders	Pebbles	0	2	Undercut	0	2	60	Improved grassland
14/09/2006	Near Little Mossy	Heltondale	348985	521012	300	14	1	0	1	2	0	0	Cobbles	Boulders	Pebbles	0	0	Undercut	1	0	0	Unimproved grassland
18/07/2006	Lowgill	Upper Eden	374595	515400	0	19	0	3	3	0	0	0	Pebbles	Gravel	Sand	1	2	None	0	0	90	Woodland
18/07/2006	Ploughlands	Upper Eden	375036	513804	171	22	0	3	0	0	0	0	Gravel	Pebbles	Sand	1	1	Poached	1	0	0	Improved grassland
18/07/2006	Ploughlands	Upper Eden	375259	513001	168	24	0	0	0	0	0	0	Pebbles	Gravel	Sand	1	2	Poached	1	0	0	Improved grassland
19/07/2006	Popping/Sturdy beck	Belah	381147	511417	171	23	0	0	0	0	0	0	Cobbles	Pebbles	Silt/mud	1	1	Poached	1	0	5	Improved grassland
19/07/2006	Howgill	Upper Eden	377217	510190	149	20	5	12	17	0	1	2	Pebbles	Gravel	Silt/mud	1	1	Poached	1	0	60	Improved grassland
19/07/2006	Howgill/Sandwath	Upper Eden	376687	509449	152	18	0	0	0	1	0	0	Gravel	Pebbles	Sand	1	1	Poached	1	0	5	Improved grassland
19/07/2006	Blacksyke	Upper Eden	372674	515787	168	14	0	1	1	0	0	0	Cobbles	Pebbles	Gravel	1	2	Poached	1	1	100	Improved grassland
19/07/2006	Mill Beck	Upper Eden	378518	510389	168	16	1	0	1	1	0	0	Pebbles	Gravel		1	2	None	0	0	60	Park/garden
19/07/2006	Popping beck	Belah	379960	511780	170	19	0	0	0	0	0	0	Pebbles	Cobbles	Silt/mud	1	0	Poached	1	0	20	Improved grassland
20/07/2006	Eden Ranunc plant site	Upper Eden	368272	520618	126	21	0	36	36	0	0	8	Cobbles	Boulders	Pebbles	0	10	None	0	0	10	Woodland
20/07/2006	Helm	Helm	370914	516922	130	21	0	4	0	0	0	0	Cobbles	Pebbles	Gravel	0	2	None	0	3	0	Improved grassland
20/07/2006	Watermillock Sike	Helm	370526	514010	180	22	0	0	0	0	0	0	Pebbles	Cobbles	Gravel	1	1	None	1	0	5	Improved grassland
10/08/2006	Upper Rampsgill	Rampsgill	343913	516005	236	12	58	0	58	1	0	10	Cobbles	Pebbles	Gravel	0	2	Undercut	1	0	5	Unimproved grassland
10/08/2006	Melbeck	Rampsgill	343694	516030	236	13	27	0	27	1	0	6	Cobbles	Boulders	Pebbles	0	1	Undercut	1	0	50	Unimproved grassland
10/08/2006	Fusedale beck	Ullswater	344811	519027	200	14	77	0	77	1	0	25	Pebbles	Cobbles	Gravel	0	2	Undercut	0	0	0	Unimproved grassland
10/08/2006	Fusedale	Ullswater	344249	519770	196	13	10	2	12	0	0	3	Cobbles	Pebbles	Gravel	0	3	None	1	0	80	Improved grassland
10/08/2006	Aik beck	Ullswater	346888	523564	162	15	34	0	34	4	0	8	Cobbles	Pebbles	Gravel	0	2	None	0	1	70	Improved grassland
10/08/2006	Aik beck	Ullswater	347218	523050	163	14	9	0	9	2	0	3	Boulders	Cobbles	Pebbles	0	2	Undercut	0	0	80	Park/garden
11/08/2006	Swindale U/S top weir	Swindale	380245	515176	221	11	3	0	3	6	0	2	Boulders	Cobbles	Pebbles	0	4	Earth cliff	1	0	70	Improved grassland
11/08/2006	Swindale D/S top weit	Swindale	380200	515116	221	11	8	0	8	3	0	3	Boulders	Cobbles	Pebbles	0	3	Earth cliff	1	0	80	Improved grassland
11/08/2006	Deadmands sike	Swindale	382331	518891	446	12	0	0	0	0	0	0	Gravel	Boulders	Cobbles	1	1	Undercut	1	0	0	Unimproved grassland
11/08/2006	Swindale head u/s waterfall	Swindale	381496	518780	447	13	1	0	1	4	0	0	Boulders	Cobbles	Bedrock	0	2	Undercut	1	0	0	Unimproved grassland
11/08/2006	Swindale d/s waterfall	Swindale	381828	518228	437	12	10	0	10	3	0	4	Cobbles	Boulders	Pebbles	0	2	None	1	0	0	Unimproved grassland
11/08/2006	Swindale at Great Musgrave	Swindale	377745	513700	158	15	0	25	25	0	1	5	Cobbles	Pebbles	Gravel	0	4	Earth cliff	0	3	0	Improved grassland
30/08/2006	Kiln beck	Raven	360306	542978	231	13	0	0	0	2	0	0	Cobbles	Boulders		0	1	Poached	1	0	10	Improved grassland
30/08/2006	Harberry beck	Raven	359548	543248	206	14	0	0	0	0	0	0	Bedrock	Cobbles	Sand	1	1	Earth cliff	1	0	80	Improved grassland
30/08/2006	North Gill	Eden Lower	352452	547940	110	14	0	0	0	1	0	0	Cobbles	Bedrock	Pebbles	1	1	Undercut	0	0	90	Unimproved grassland
31/08/2006	Irthing	Irthing	349260	560251	66	15	0	35	35	0	0	6	Cobbles	Pebbles	Gravel	0	8	None	0	2	5	Woodland
31/08/2006	Irthing	Irthing	350441	561086	46	15	0	18	18	0	1	6	Cobbles	Pebbles	Gravel	0	8	Earth cliff	1	2	0	Improved grassland
31/08/2006	Brampton beck	Irthing	351087	561009	50	15	0	0	0	0	0	0	Gravel	Sand		1	1	Poached	1	0	0	Improved grassland
31/08/2006	Poltross burn	Irthing	362325	564852	173	14	0	0	0	0	0	0	Sand	Gravel	Cobbles	1	2	Undercut	1	0	30	Improved grassland
31/08/2006	Harrow beck	Irthing	361659	566982	151	16	0	0	0	0	0	0	Bedrock	Cobbles	Pebbles	1	1	Undercut	0	0	0	Unimproved grassland
31/08/2006	Irthing	Irthing	358461	564465	147	18	0	8	8	0	1	5	Boulders	Cobbles	Pebbles	0	8	None	0	5	10	Improved grassland
31/08/2006	Pottscleugh beck	Irthing	358212	563111	142	19	0	0	0	0	0	0	Cobbles	Pebbles	Gravel	1	1	Earth cliff	1	0	30	Improved grassland
31/08/2006	Upper end Milburn	Milburn	357357	559960	168	18	5	0	5	0	0	0	Cobbles	Pebbles	Gravel	0	0	Undercut	1	0	0	Improved grassland
01/09/2006	Hawkney beck	Cairn	350892	556390	61	14	0	0	0	0	0	0	Cobbles	Bedrock	Sand	0	1	None	0	0	90	Woodland

Appendix 2: Land cover classification scheme based on the UK Biodiversity Action Plan (Source: CEH Land Cover Map 2000).

LCM Target class	LCM Subclasses (Level-2)	Variants (Level-3)
Sea / Estuary	Sea / Estuary	sea
Water (inland)	Water (inland)	water (inland)
Littoral rock and sediment	Littoral rock	rock
		rock with algae
	Littoral sediment	mud
		sand
		sand with algae
	Saltmarsh	saltmarsh
		saltmarsh (grazed)
Supra-littoral rock and sediment	Supra-littoral rock	rock
	Supra-littoral sediment	shingle (vegetated)
		shingle
		dune
		dune shrubs
Bog	Bogs (deep peat)	bog (shrub)
		bog (grass/shrub)
		bog (grass/herb)
		bog (undifferentiated)
Dwarf shrub heath (wet / dry)	Dense dwarf shrub heath	dense (ericaceous)
		gorse
	Open dwarf shrub heath	open
Montane habitats	Montane habitats	montane
Broad leaved / mixed woodland	Broad leaved / mixed woodland	deciduous
		mixed
		open birch
		scrub
Coniferous woodland	Coniferous woodland	conifers
		felled
		new plantation
Arable and horticultural	Arable cereals	barley
		maize
		oats
		wheat
		cereal (spring)
		cereal (winter)
	Arable horticulture	arable bare ground
		carrots
		field beans
		horticulture
		linseed
		potatoes
		peas
		oilseed rape
		sugar beet
		unknown
		mustard
		non-cereal (spring)
	Non-rotational arable and horticulture	orchard
		arable grass (ley)
		setaside (bare)
		setaside (undifferentiated)

LCM Target class	LCM Subclasses (Level-2)	Variants (Level-3)
Improved grassland	Improved grassland	intensive grass (hay/ silage cut) grazing marsh
Rough and semi-natural neutral and calcareous grasslands	Setaside grass	grass setaside
	Neutral grass	neutral grass (rough) neutral grass (grazed)
	Calcareous grass	calcareous (rough) calcareous (grazed)
Acid grass and bracken	Acid grass	acid acid (rough) acid with *Juncus* acid *Nardus/Festuca/Molinia*
	Bracken	bracken
Fen, marsh and swamp	11. Fen, marsh and swamp	swamp fen/marsh fen willow
Built up areas, gardens	Suburban/rural developed	suburban/rural developed
	Continuous Urban	urban residential/commercial urban industrial
Inland Bare Ground	Inland Bare Ground	despoiled semi-natural

Appendix 3: Water sampling protocol used by all samplers

1. Rinse bottle & cap thoroughly 3 times with stream water at site before sample collection. Empty downstream.
2. When filling the bottle always make sure you are stood downstream so as not to disturb bottom sediments.
3. Always hold the bottle facing upstream when collecting the sample making sure your hands are downstream of the opening. This is to prevent salts washing off your hands into the sample.
4. Remove bottle from stream before fastening cap again to prevent salts washing off your hands.
5. Always fill the bottle to the rim to ensure as little air as possible is trapped within the bottle. Top up using cap if necessary in shallow streams.
6. Always collect samples from flowing water, preferably from a riffle within the main flow of the stream. If water is not flowing and is stagnant please make a note on the record sheet.
7. Try to avoid collecting large amounts of particulate matter such as sediment and pieces of vegetation.
8. During transport try to keep samples as cool as possible and out of direct sunlight e.g. in boot of car.
9. If you collect any additional samples please label with stickers provided, mark location on map and record in sample table.
10. If any sites are inaccessible or dry do not sample just make a note in the table.

Appendix 4: Rotated component matrices of habitat factors extracted by PCA at the area-scale with an eigenvalue >1. The factor which each habitat control is most strongly related is highlighted in bold

Area: 1 Tyne Gap

	Factor Number					
	1	2	3	4	5	6
Dominant substrate	.017	-.154	.084	**.847**	-.016	.324
Gravel presence	.248	.188	.046	**.788**	.169	-.233
Siltation	.101	-.128	**.701**	-.106	.024	.194
Width	-.253	-.381	**.662**	-.025	-.063	-.261
Slope	-.238	-.227	**-.525**	-.331	.363	-.060
Biotope	-.367	-.111	**-.630**	-.107	.188	.056
Erosion presence	.394	**.575**	-.072	-.007	.327	.363
Stock access	-.222	.224	.061	.053	-.090	**.849**
Erosion on both banks	.022	**.961**	-.148	-.037	-.014	.100
Erosion severity	.085	**.934**	-.130	.080	.059	.021
Stock erosion	-.048	.022	-.138	.091	**.892**	-.092
Fluvial erosion	.100	**.927**	-.087	.051	-.216	.049
% overhead cover	-.418	**.640**	.404	-.102	.032	.309
Tunnelled vegetation	.213	**-.512**	-.436	-.098	-.194	.189
Riparian land cover	.341	-.098	**.520**	.157	.425	.259
Catchment-channel hydrological connectivity risk						
Unclassified	**.877**	-.059	.108	.156	-.101	-.137
Classified	**.825**	.061	.145	-.081	.158	.063
Optimised	**.863**	.223	-.006	.182	-.010	-.097
Catchment-channel hydrological connectivity risk weighted by land cover						
Unclassified	**.667**	-.187	.505	.339	-.155	-.085
Classified	**.610**	-.142	.590	.188	-.162	.002
Optimised	**.599**	.121	-.014	.590	.015	-.071

Area 2: Pennine Becks

	Factor Number					
	1	2	3	4	5	6
Dominant substrate	.118	.032	.060	-.203	**.749**	.154
Gravel presence	.251	.129	-.008	-.165	.391	**.696**
Siltation	.316	.139	-.120	.183	.200	**-.594**
Width	-.361	.031	.051	**-.597**	.057	.224
Slope	-.111	-.112	.006	**.711**	-.005	-.084
Biotope	-.118	.031	-.119	**.651**	-.120	.594
Erosion presence	.056	**.728**	.020	.087	-.161	-.017
Stock access	.013	**.677**	.059	-.077	-.150	.095
Erosion on both banks	-.075	**.844**	.317	.043	.163	-.121
Erosion severity	-.106	**.863**	.033	-.102	.139	-.054
Stock erosion	-.019	.572	**-.719**	-.036	-.002	-.035
Fluvial erosion	-.126	.382	**.862**	.012	.117	-.053
% overhead cover	.017	.271	**.563**	-.218	-.401	.163
Tunnelled vegetation	.248	-.325	-.107	.225	**.660**	-.253
Riparian land cover	-.051	**.606**	-.105	-.377	-.145	.145
Catchment-channel hydrological connectivity risk						
Unclassified	**.908**	-.084	-.035	-.073	.124	-.023
Classified	**.882**	.024	-.026	-.029	-.027	.081
Optimised	**.816**	-.065	.064	.139	-.047	-.025
Catchment-channel hydrological connectivity risk weighted by land cover						
Unclassified	**.920**	-.033	-.053	-.091	.203	-.022
Classified	**.825**	.035	-.027	-.083	.166	.007
Optimised	**.778**	-.042	-.053	.165	.031	-.160

Area 3: Orton and Howgill Fells

	Factor Number					
	1	2	3	4	5	6
Dominant substrate	-.095	-.001	.284	**.623**	.449	.086
Gravel presence	.071	.074	.060	**.744**	-.150	.045
Siltation	-.195	.238	**.446**	.221	-.291	.003
Width	.112	-.208	.064	**-.612**	.111	.469
Slope	-.258	-.116	**-.708**	.079	-.043	.228
Biotope	-.199	-.010	**-.854**	-.065	.001	-.070
Erosion presence	-.279	**.572**	.251	.084	.162	-.096
Stock access	-.151	**.710**	.018	.208	-.041	-.104
Erosion on both banks	.044	**.934**	.055	.055	.055	-.065
Erosion severity	.092	**.934**	.119	.006	.065	.010
Stock erosion	.148	**.778**	-.051	-.094	-.085	.510
Fluvial erosion	-.090	.106	.215	.085	.172	**-.788**
% overhead cover	.129	.166	.182	-.308	**.639**	-.062
Tunnelled vegetation	-.044	-.008	.182	-.082	**-.850**	.066
Riparian land cover	.083	.170	.387	.140	.078	**.469**
Catchment-channel hydrological connectivity risk						
Unclassified	**.975**	-.008	.058	-.011	.002	-.032
Classified	**.926**	.038	.023	-.028	-.004	-.026
Optimised	**.623**	-.224	.144	-.461	-.009	.219
Catchment-channel hydrological connectivity risk weighted by land cover						
Unclassified	**.961**	.031	.056	.072	.062	.015
Classified	**.863**	-.030	.138	-.036	.079	.182
Optimised	**.837**	-.061	.091	-.043	.074	.077

Area 4: Ullswater & Lowther Valley

	Factor Number					
	1	2	3	4	5	6
Dominant substrate	-.171	.078	.003	-.179	.177	**.805**
Gravel presence	.010	-.110	-.022	.015	.042	**.805**
Siltation	-.066	-.077	-.148	.051	**.680**	.161
Width	.317	.026	.123	**-.705**	-.325	-.015
Slope	-.235	.122	-.140	**.445**	-.443	-.241
Biotope	.098	.016	.119	**.682**	-.056	-.096
Erosion presence	-.061	.407	-.030	.131	**.557**	.078
Stock access	.053	**.653**	.060	-.266	.084	-.038
Erosion on both banks	-.062	**.947**	-.104	.069	.071	-.004
Erosion severity	-.032	**.931**	.065	.051	.133	-.111
Stock erosion	.016	.354	.489	-.235	**.553**	-.199
Fluvial erosion	-.008	**.789**	-.311	.153	-.330	.101
% overhead cover	.062	.305	**-.756**	-.122	.198	.200
Tunnelled vegetation	.114	-.294	**.591**	.374	-.140	.032
Riparian land cover	.148	.181	**.593**	-.175	.100	.153
Catchment-channel hydrological connectivity risk						
Unclassified	**.937**	-.051	.038	-.123	-.024	-.037
Classified	**.870**	-.006	-.042	.222	.107	-.101
Optimised	**-.921**	.011	-.105	.108	.066	.002
Catchment-channel hydrological connectivity risk weighted by land cover						
Unclassified	**.964**	-.019	.031	-.062	-.039	-.097
Classified	**.893**	.015	-.008	.083	.059	-.070
Optimised	**-.854**	-.002	-.151	.222	.158	-.150

Area 5 – Caldew & Petteril Rivers

	Factor Number					
	1	2	3	4	5	6
Dominant substrate	.150	.061	.069	.144	-.037	**.788**
Gravel presence	**.397**	.342	.160	-.159	.246	.258
Siltation	**.751**	.311	.076	.081	.056	.018
Width	-.215	-.414	**-.676**	-.036	-.232	-.042
Slope	-.253	.088	**-.867**	.030	-.101	.089
Biotope	-.168	-.068	**-.484**	-.108	.185	-.384
Erosion presence	.227	**.603**	.159	.313	.032	-.144
Stock access	.010	.334	.120	.432	-.066	**-.641**
Erosion on both banks	-.090	.169	.068	.486	**.819**	.018
Erosion severity	.142	.162	.077	**.785**	.519	.045
Stock erosion	.201	.021	.184	**.897**	-.070	.028
Fluvial erosion	-.159	.108	-.039	-.126	**.938**	-.068
% overhead cover	-.258	**.791**	-.009	-.006	.197	-.101
Tunnelled vegetation	-.190	**-.818**	.220	-.004	-.010	-.099
Riparian land cover	-.013	-.014	**.679**	.367	-.135	.044
Catchment-channel hydrological connectivity risk						
Unclassified	**.964**	.141	.105	.062	-.098	.075
Classified	**.941**	-.077	.144	.126	-.137	.111
Optimised	**.580**	.503	.379	.175	.115	-.050
Catchment-channel hydrological connectivity risk weighted by land cover						
Unclassified	**.962**	.188	.088	.035	-.017	.038
Classified	**.941**	-.077	.144	.126	-.137	.111
Optimised	.351	**.754**	.265	.052	.074	.038

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